Managing our land for multiple ecosystem services

Verhagen, W.

2019

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Identifying priority areas and actions to maintain ecosystem services across Europe

Willem Verhagen

VRIJE UNIVERSITEIT

MANAGING OUR LAND FOR MULTIPLE ECOSYSTEM SERVICES

Identifying priority areas and actions to maintain ecosystem services across Europe

ACADEMISCH PROEFSCHRIFT

ter verkrijging van de graad Doctor of Philosophy
aan de Vrije Universiteit Amsterdam,
op gezag van de rector magnificus
prof.dr. V. Subramaniam,
in het openbaar te verdedigen
ten overstaan van de promotiecommissie
van de Faculteit der Bètawetenschappen
op vrijdag 14 juni 2019 om 13.45 uur
in de aula van de universiteit,
De Boelelaan 1105

door

Willem Verhagen

geboren te Helmond

promotor: prof.dr.ir. P.H. Verburg

copromotor: dr. A.J.A. van Teeffelen

Promotiecommissie

prof.dr. R. Aerts
dr. P.J.H. van Beukering
prof.dr. A. Grêt-Regamey
dr. G.B.M. Pedroli
dr. P.A. Verweij

This research was conducted under the auspices of the Graduate School for Socio-Economic and Natural Sciences of the Environment (SENSE)

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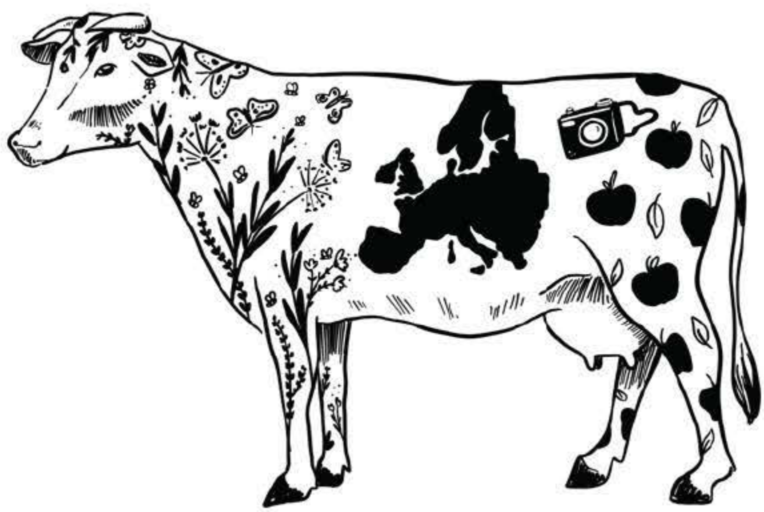


1. General Introduction - maintaining ecosystem services across landscapes in Europe

1.1 Integrating ecosystem services into decision making for conservation and restoration of land

Ecosystem services are the benefits human societies obtain from nature. These ecosystem services (ESs) are essential components to sustain human life and well-being (MEA, 2005). ESs play a role in many human-nature relations, such as walking your dog in the city park, pollination of cacao trees by midges or carbon storage in peat bogs. ESs can be classified into three categories: (1) provisioning services, ESs that provide a good such as drinking water or wild berries; (2) regulating services, ESs that regulate the quality of air, water or soils such as air purification, flood control and sediment retention; and (3) cultural services, ESs that provide cultural value to human society such as recreational activities and inspiration for art. The long-term provision of a range of ESs depends on well-functioning ecosystems.

However, the proper functioning of ecosystems is threatened by human use of the land (Rockström et al., 2009; Steffen et al., 2015). Managing the available land for a narrow set of, often provisioning, services has resulted in the (unintended) decline of biodiversity and many regulating services (MEA, 2005; Bennett et al., 2009; Newbold et al., 2015). In the near future, pressures on land will increase further given expected rises in demand for all types of ESs (Alcamo et al., 2005; Stürck et al., 2015b). Therefore, the human use of the land needs to be adapted to ensure the provision of a diverse set of ESs within and across landscapes.

The concept of ESs is increasingly adopted in policies on water, land and natural resource management. These policies mostly focus on maintaining ESs over time through the conservation and restoration of land. At the global level, the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) was launched with the aim to foster international collaboration on maintaining biodiversity and its related services. ESs were included alongside biodiversity conservation as criteria for the further expansion of protected area networks and the restoration of natural areas (Convention on Biological Diversity, 2010). At the level of the European Union (EU), recent policies are more likely to include targets for maintaining ESs than before (Bouwma et al., 2018). The European Commission has set the target to halt the loss of ESs by 2020 (European Commission, 2011). Thus, there are several policy frameworks in place that aim to maintain ESs over time. To realise these ambitions a next step is the implementation of ESs into decision making on where and how to manage land.

An important component of supporting decision making is quantifying the spatial variation of ESs; in other words mapping ESs (Maes et al., 2012). Both ES capacity and ES demand tend to vary across locations and scales. A wide variety of approaches exist to quantify this spatial variation (Seppelt et al., 2011a; Maes et al., 2012; Martínez-Harms and Balvanera, 2012). In the last two decades researchers have reviewed methods to map ESs, assessed the quality of ES maps and developed shared indicators for mapping ESs (Burkhard et al., 2009; Eigenbrod et al., 2010; Egoh et al., 2012; Martínez-Harms and Balvanera, 2012; Crossman et al., 2013; Schulp et al., 2014a; Verhagen et al., 2014; Maes et al., 2016). ES maps are now available for a diverse set of ESs in many regions in the world. Moreover, ES maps are increasingly being integrated into decision-support tools designed to provide advice on how to manage land for ESs (Grêt-Regamey et al., 2017).

So how can these maps be used to support land management for ESs? I distinguish three broad ways of using ES maps for decision making (Verhagen et al., 2014). First, ES maps can be used for the assessment of individual ESs, i.e. to evaluate the (current) state of ESs. This is a step that the EU indeed takes as part of the Biodiversity Strategy: all Member States are required to map their ESs by 2020 (European Commission, 2011). The ES maps mainly have an informing role by visualising ESs distributions over space. Consecutive assessments over time show the magnitude and location of change in ESs, which help to identify underlying driving forces of change. Such assessments are a necessary first step towards including ESs into decision making.

Second, assessments of multiple ESs can be combined to assess trade-offs and synergies. This gives an indication of which ESs occur in conjunction and whether the relations between ESs are uniform across space and time. An example is the identification of ES bundles, a set of associated services that usually occurs together in time or space (Raudsepp-Hearne et al., 2010). The concept of ESs aims to be cross-sectoral by integrating disciplines on water management, natural resource management, biodiversity protection and so forth (Primmer and Furman, 2012). This implies that policies on natural resources need to meet multiple, sometimes competing, goals. To support these policies, an interesting first step is to use ES maps to quantify and visualise trade-offs or synergies between different ESs. A second step is to compare trade-offs between ESs resulting from different land management decisions. Linking trade-offs between ESs to land management decisions allows decision makers to minimize trade-offs.

Third, ES assessments can be used for prioritization: the identification of priority areas or priority management actions for ESs. Funds and land for conserving and restoring ESs are intrinsically limited. Identifying where (priority area) and how (priority action) to manage land most effectively to achieve objectives can therefore be helpful, or even necessary to support decision making. An example are the global Aichi target which aim to protect by 2020 "at least 17 percent of terrestrial and inland water, especially areas of particular importance for biodiversity and ecosystem services" (Convention on Biological Diversity, 2010). Here the target is to protect part of the land with the goal to most effectively maintain biodiversity and ESs, or in other words to identify priority areas for protection of biodiversity and ESs. Prioritization approaches were originally developed for biodiversity conservation and are now adopted to support decisions on where and how to manage land for ESs. Priority areas can be identified based on overlays of individual ES maps (Gimona and Horst, 2007; Davids et al., 2016). An alternative is to use ES maps as an input into (partial) optimization approaches to achieve a specified conservation targets (Chan et al., 2006; Moilanen et al., 2009; Egoh et al., 2011).

In this thesis I address all of the before mentioned ways in which ESs maps can be used to support decision making. These include assessments of ESs (over time), trade-offs between ESs and prioritization of areas and actions. The primary focus of this thesis is on prioritizing areas and actions to maintain ESs within Europe. Prioritization approaches can directly support decision making on where and how to conserve and restore land for ESs, allocate land and funds effectively given policy objectives and assess trade-offs between ESs based on multiple, alternative land management options.

Decision-support tools to identify priority areas and actions rely heavily on ES maps as input. Therefore, the quality of the decision support is limited by the quality of the maps used as input. There are multiple methods available to quantify the spatial variation in ESs. A critical reflection on ESs maps is a necessity prior to utilizing these maps to support decision making (Schulp et al., 2014a). In the next section (1.2), I review current approaches to map and assess ESs.

1.2 Quantifying and mapping ecosystem services

1.2.1 Linking land use and ecosystem services

The number of studies mapping ESs has steadily increased in the past decades. Much effort has gone into reviewing methods to map ESs (Egoh et al., 2012; Martínez-Harms and Balvanera, 2012; Wolff et al., 2015) and the development of common indicators (Maes et al., 2016). Although more standardized approaches become available, a comparison of four different methods to quantify ESs for Europe showed considerable differences between ES maps (Schulp et al., 2014a). Currently no standards exist to independently assess the quality of ES maps. At larger spatial extents it becomes difficult to ground truth ES maps with observational data. Irrespective of the spatial extent, the wealth of ES maps makes validation costly and highly time consuming. Indicators used to map ESs need to be able to capture the dominant processes that determine variation in a particular ES over time and space. However, the quality of an ES map should also be evaluated based on the intended use of the map. The suitability of a method to map ESs partly depends on the intended application, implying that not all ES maps are suitable for all types of decision support (Verhagen et al., 2014). Therefore, to support decision making indicators used to map ESs also need to relate to the dominant processes in which humans change land management.

Humans change and manage land in diverse ways. Human societies have predominantly altered land through the large scale clearing of natural vegetation (Klein Goldewijk, 2001; Ramankutty et al., 2008; Ellis et al., 2013), the simplification of landscape structure (Fahrig, 2003; Bianchi et al., 2006) and the intensification of land management (Foley et al., 2011). Together, changes in the composition of land cover, the configuration of land cover and the management intensity are a dominant force in how humans affect landscapes. Consequently, changes in these three landscape characteristics threaten biodiversity, affect processes related to the hydrological cycle or nutrient cycling and undermine the sustainable supply of multiple, diverse ESs (MEA, 2005; Rockström et al., 2009; Pereira et al., 2010).

In this thesis I distinguish between landscape composition, landscape configuration and management intensity. Landscape composition refers to the amount of land cover types within a landscape, whereas landscape configuration refers to the spatial arrangement, or structure, of these land cover types within a landscape (Fahrig et al., 2011). For example, two landscapes consist of equal areas of forest and agricultural land (composition). The first landscape consists of two large areas of forest and agricultural land, whereas the second landscape consists of multiple smaller areas of forest and agricultural land (configuration). Within these landscapes part of the forest area is strictly protected for biodiversity whereas another part is a plantation forest for wood production (management intensity). Both composition and configuration can be partly derived from land cover datasets. Information on management intensity cannot be derived from land cover datasets. Combined, I refer to them in this thesis as the three "main" facets of land.

Landscape composition, landscape configuration and management intensity affect the mapping of ESs. These three facets of land also affect the identification of priority areas and actions for ESs. Explicitly considering all three facets provides opportunities to design and manage landscapes capable of addressing global needs for ESs based in local realities (Verburg et al., 2013, 2016). It provides decision makers with the opportunity to assess a wider set of landscape management options to conserve and restore ESs. Therefore, identifying priority areas and actions for ESs needs to consider landscape composition, landscape configuration and management intensity.

Land cover and land use datasets are most commonly used to map ESs. Especially at larger extents land cover datasets are important data sources for ES quantification, given the limited availability of data on other variables at this extent. Land cover data are most often used to link landscape composition and ES values. A review of ES case studies showed that around half (49%) of the studies that map ESs rely on a relation with landscape composition (Lautenbach et al., 2019) At its most basic form, land cover datasets are directly transferred to an ES quantity for capacity and demand, using so-called lookup tables (e.g. Burkhard et al., 2014). This is the most common applied method to quantify the spatial variation in ESs (Seppelt et al., 2011b, 2011a; Lautenbach et al., 2019). Although commonly used, land cover proxies are heavily criticized, because it assumes a uniform level of ESs per land cover type across the spatial extent of a study region. Comparisons to field measurements showed that this is an oversimplification resulting in erroneous quantification of spatial variation in ESs (Eigenbrod et al., 2010; Roussel et al., 2017). Alternative methods to map ESs include the use of process-based models, causal relationships or extrapolation of primary data (Martínez-Harms and Balvanera, 2012; Lavorel et al., 2017). Partly these methods still rely on land cover datasets for parameterization and for the extrapolation of data. Overall, many studies quantifying ESs account for effects of landscape composition and changes therein on ESs.

Information on landscape configuration and management intensity is less often used to map ESs. A review of ES case studies showed that for those studies mapping ESs 17% accounted for landscape configuration and 20% accounted for management intensity (Lautenbach et al., 2019). Partly, this is because there is limited empirical evidence for how these two land facets affect ESs. Moreover, there is limited availability of spatial datasets on landscape configuration and management intensity (Kuemmerle et al., 2013). This hampers the integration of landscape configuration and management intensity in mapping ESs and consequently into the prioritization of areas and actions for ESs.

However, in recent years some important improvements have been made. Currently, European wide datasets exist on the spatial distribution of linear elements in agricultural landscapes (van der Zanden et al., 2013). Also data on the spatial variation of management intensity across European agricultural lands and forests is now available (Temme and Verburg, 2011; Hengeveld et al., 2012). These type of datasets have now also been used to map several ESs across Europe (Schulp et al., 2014b; Stürck et al., 2014; Panagos et al., 2015). The increasing availability of these datasets provides opportunities for further integrating landscape configuration and management intensity in mapping ESs. Prior to integrating these factors in the mapping of ESs, we need to know how ESs are affected by landscape configuration and management intensity. In section 1.2.1 I provide an overview of the knowledge base of how landscape configuration affects ESs. In section 1.2.2 I do the same for management intensity.

1.2.2 Effects of landscape configuration on ESs

Landscape ecology has a long history in studying the effects of landscape configuration on biodiversity and ecosystem processes (Turner, 1989). Landscape configuration can be assessed at the patch and the landscape level. It describes the spatial arrangement of land cover types, the connectivity of forest areas, the amount of edges between forests and agricultural land or the network of hedgerows through agricultural fields. Landscape configuration is not commonly accounted for in the quantification of ESs.

There is evidence that landscape configuration affects single ESs. A theoretical model quantified losses in pollination following natural land cover loss. The losses in pollination intensified with increasing fragmentation of remaining habitats (Mitchell et al., 2015a). A field study on sediment export observed a two-to-five fold variation in sediment export, whilst keeping the total amount of habitat conversion constant (Chaplin-Kramer et al., 2016). Hence, in both studies changes in ESs could only be partly explained by changes in landscape composition alone and also depend on changes in the configuration of land cover.

Several studies have also looked at the effect of landscape configuration on multiple ESs. These studies showed that each ES has a unique response to changes in landscape configuration (Mitchell et al., 2014a; Lamy et al., 2016). For example, in a case study increasing forest fragmentation did not significantly impact all ESs. Moreover, those ESs that were affected showed differing responses to increasing forest fragmentation (Mitchell et al., 2014a). These findings suggest that changes in ES capacity were affected by both landscape composition and landscape configuration. Each ES had a unique response to changes in landscape configuration.

Besides some individual field studies, we still have limited knowledge on how landscape configuration affects ESs. A review found that only 15 studies in the field of ESs provide empirical evidence for a relation between landscape configuration and ESs (Mitchell et al., 2013). These studies mostly focused on pollination. In these studies decreasing connectivity in landscapes reduced ES capacity (Mitchell et al., 2013). However, this is a very limited knowledge base to relate effects of landscape configuration to multiple ESs. So far, we are missing an overview of which, how and how strongly ESs are affected by landscape configuration.

All previous studies focused on how landscape configuration affects the capacity of a landscape to provide a service. Changes in landscape configuration affect both ES capacity and ES flow. Often the areas with high ES capacity are not the areas where people use or need particular ESs. For example, forested areas in an upstream catchment can supply flood regulation to populated areas in downstream catchments. The spatial connections between areas of ES demand and capacity are referred to as ES flows (Serna-Chavez et al., 2014). These ES flows can range from local to global connections. Accounting for the spatial configuration of ES demand and ES capacity areas better links human societies to valuable ecosystems.

Changes in landscape configuration also affect the ES flow. However, the evidence for effects of landscape configuration on ES flow is even more limited. Researchers have argued that landscape fragmentation can reduce ES capacity while at the same time increase the flow of those very same ES (Mitchell et al., 2015b). These conceptual differences highlight the importance of accounting for the effects of landscape configuration on ES capacity and ES flow separately. So far there has not been much empirical evidence to support effects of landscape configuration on ES flow.

Landscape configuration is relevant for decision making on how to manage landscapes for ESs. Researchers have argued that actively changing the landscape configuration can provide interesting opportunities for restoring ESs (Turner et al., 2013). Especially the restoration and protection of landscape elements such as ponds, tree lines or hedges in agricultural areas holds great promise (Jones et al., 2013). An example of this is given by a study on forest restoration. In this study forest restoration decreased nitrogen and sediment export, but the actual achieved reduction strongly depended on the locations of forest restoration (Barnett et al., 2016).

Taken together, accounting for configuration has the potential to improve the assessment of ESs. It also widens the toolbox to manage landscapes for ESs. Currently we are missing an overview of how landscape configuration affects the level and spatial variation of multiple ESs. This is a crucial first step prior to integrating landscape configuration into prioritization approaches aiming to identify priority areas and actions to manage landscapes for ESs.

1.2.3 Management intensity

Globally, land management in many areas is optimized for the provision of a single ESs, namely food, feed or timber. This singular focus has negative consequences for biodiversity and non-provisioning ESs (Bennett et al., 2009; Seppelt et al., 2016). As crop systems worldwide also depend on ESs and biodiversity, e.g. through soil quality or pollination, a focus on management practices for multiple ESs is necessary (Tscharntke et al., 2012). However, making crop systems locally more biodiverse and ES friendly might result in biodiversity loss at larger scales given the need for additional cropland to meet rising demand for food (Green et al., 2005; Phalan et al., 2011b). It is therefore crucial to understand how changes in management intensity affect multiple ESs and the local and global sustainability of multifunctional landscapes (Benton et al., 2018).

ES capacity is affected by management intensity directly as well as indirectly, through changes in biodiversity (van Oudenhoven et al., 2012). A meta-analysis showed that increasing land management intensity reduced species abundance in forest, croplands and grasslands (Newbold et al., 2015). In a field study, changing management intensity of grasslands had both direct and indirect effects on ES capacity (Allan et al., 2015). Thus, there is evidence that management intensity affects ESs through changes in biodiversity.

There is also case study evidence for direct effects of management intensity on ESs. Just as with landscape configuration, ESs respond in unique ways to management intensification. In the UK, increasing the intensity of forest management induced strong trade-offs between wood production and other ESs (Sing et al., 2018). In vineyards, a decrease in management intensity showed strong positive unique increases in several ESs, especially erosion reduction (Winter et al., 2018). In grasslands, differences in the degree of fertilization resulted in contrasting trends in ES capacity for multiple services (Schirpke et al., 2017). In total, changing management intensity did not only alter the capacity of the land to provide individual ESs but especially changed its capacity to support a combination of ESs (Allan et al., 2015; Schirpke et al., 2017). Thus, there is ample evidence that land management intensity affects ESs and the multifunctionality of landscapes.

We miss an understanding of how management intensity affects relations between multiple ESs and biodiversity. Meta-studies that generalize findings focus on single ESs, thereby ignoring possible trade-offs between ESs. How management intensity varies globally and what the consequences thereof are for the trade-offs between biodiversity and multiple services in production landscapes remains largely unknown. Most likely a diversity of management approaches is required to maintain multiple ESs across croplands, forests and grassland systems (Sing et al., 2018) Better knowledge on the link between management intensity, biodiversity, production and the associated feedbacks is especially important for the design of multifunctional landscapes. Multifunctional landscapes are capable of delivering multiple ESs simultaneously, thereby balancing between multiple competing objectives (Tscharntke et al., 2012; Fischer et al., 2014; Seppelt et al., 2016).

Although empirical evidence exists for effects of management intensity on ESs, it requires further integration in the mapping of ESs. A difficulty is the lack of datasets on the spatial variation of management intensity (Kuemmerle et al., 2013; Erb et al., 2017). Unlike land cover data, spatial explicit data on management intensity is often only available for smaller spatial extents or in an aggregated coarser resolution. With time however, improvements are being made in mapping management intensity variation at larger spatial extents (Temme and Verburg, 2011; Siebert et al., 2013; Overmars et al., 2014; Fritz et al., 2015; Bouwman et al., 2017; Wu et al., 2018). These improvements provide the opportunity to better account for variation in management intensity in the mapping of ESs. Some studies have already done so (Stürck and Verburg, 2017). The large majority of ES assessments, however, have not incorporated variation in management intensity.

Accounting for management intensity is essential for decisions on where and how to manage land for ESs. Many conservation and restoration efforts focus on changes in land management practices instead of changes in land cover. For example, researchers have studied how biodiversity and ESs respond to adopting organic or biodiversity-friendly farm management (Winqvist et al., 2011; Kremen and Miles, 2012; Duru et al., 2015). At the global level, much of the projected rise in food production will come from changes in management intensity in stead of expansion of agricultural land (Foley et al., 2011; Mueller et al., 2012; Seppelt et al., 2016; van Ittersum et al., 2016). Incorporating management intensity into ES quantification provides better insights in why ESs change over time and gives decision makers more options for maintaining and restoring ESs over time.

1.2.4 Spatial variation in ESs based on composition, configuration and management intensity

The interactions between land and ESs are multifaceted and include processes beyond changes in land cover alone. The previous sections have shown that empirical evidence exists for effects of landscape configuration and management intensity on multiple ESs. The knowledge base for these effects is still limited. To date we are missing an overview of how landscape configuration affects individual ESs. Also for management intensity we are missing an overview of how it affects multiple services and the trade-offs between those services. There is thus a need to expand this knowledge base.

The mapping of ESs is important to support decision making on where and how to manage land for ESs. Many of the methods used to map ESs account for landscape composition, but hardly do ES assessments account for all three facets of land (Lautenbach et al., 2019). Therefore, prioritization studies cannot assess the relative potential of each component for the conservation and restoration of ESs. Accounting for effects of landscape configuration and management intensity provides the opportunity to assess more management actions beyond a focus on changes in land cover only. Importantly, it provides an opportunity to improve assessments of trade-offs and ways to minimize these trade-offs, and it quantifies the threats to ESs over time beyond land cover change. Ideally accounting for all facets of lands supports decision makers in making more accurate decisions on how to manage the land for multiple ESs. In the next section (1.3) I discuss how these ES maps can be used in prioritization of areas and actions for maintaining ESs.

1.3 Prioritizing areas and actions for maintaining ESs

Several policies exist that aim to maintain and improve the level of ESs over time. These policies tend to be land based, meaning that this goal can be achieved through conservation and restoration of well-functioning ecosystems. To support the implementation of policies on restoration and conservation researchers have developed approaches that can identify priorities for the most cost-effective conservation or restoration networks (Moilanen et al., 2009). Identifying priority areas and actions is important given the manifold demands on land and limited budgets for the maintenance of biodiversity and ESs.

Prioritization can focus on identifying priority areas or priority actions. Identifying priority areas answers the questions where are the most important areas for a particular set of ESs, without answering what should be done with this area. Identifying priority actions answers the question what can best be done at a certain location. Management actions cover a wide array of decision on both whether and how to manage a particular area. Commonly identifying areas and actions are done in combination. For example, often priority areas are identified "for conservation" or priority actions are identified for specific locations in the landscape. Nonetheless, methods to identify priority areas and actions differ. In the next sections I discuss the current knowledge on prioritizing areas (1.3.1) and actions (1.3.2) for ESs.

1.3.1 Identifying priority areas for ecosystem services

Studies that identify priority areas for ESs often use ES maps as an input. How these priority areas are identified depends on the chosen methodology. Here, I distinguish between hotspot approaches and spatial conservation prioritization approaches. These are the two main approaches applied to identify priority areas for ESs.

A common approach is to identify ES hotspots, i.e. locations with a very high capacity of single and multiple ESs (Gimona and Horst, 2007; Crossman and Bryan, 2009; Davids et al., 2016; Li et al., 2017; Liu et al., 2017). ES hotspots are often identified as locations that meet or surpass a certain standard, such as the top 10% of locations that offer the highest level of ES capacity. Alternatively these approaches can identify ES coldspots (Comín et al., 2018). Hotspot approaches have been used to inform decision making. The identification of ES hotspots has been used to inform and operationalize policies on payment for ES schemes, land degradation neutrality and the incorporation of ESs into protected area networks (Wendland et al., 2010; Willemen et al., 2016; Spanò et al., 2017).

Hotspot approaches are optimal when considering a single objective (here an ES), but have limitations when multiple objectives (ESs) are considered. An issue with hotspot approaches is that these approaches ignore complementarity. Therefore, the combination of individual hotspot locations can be suboptimal, in terms of the level of conservation achieved in the targeted locations, or in terms of the resources used to meet the target. Hotspot approaches are therefore less suitable to identify priority areas for multiple ESs (Schröter and Remme, 2016)

Researchers have developed tools designed to identify priority areas for multiple objectives and explicitly deal with trade-offs between objectives. Being able to balance multiple objectives is crucial when identifying priority areas for multiple ESs, multiple species or a combination of both. These approaches are referred to as spatial conservation prioritization (Possingham et al., 2000; Wilson et al., 2007; Moilanen et al., 2009). These tools use (partial) optimization methods and the principles of concepts like complementarity and (ir)replaceability. Optimization methods are then combined with information on species distributions and habitat quality, threats to species and/or conservation costs, (Moilanen et al., 2009). Spatial prioritization approaches are specifically designed to incorporate multiple objectives (here: ESs) and to achieve a balance between competing services (Moilanen et al., 2009). A great advance of spatial prioritization approaches over hotspot identification and scoring approaches is that optimization approaches can identify an efficient set of sites and/or actions, considering multiple objectives and their distributions, potential threats and/or costs simultaneously.

Prioritization methods have been refined over the years and have been applied in diverse conservation contexts. To date, important lessons have been learned from applying spatial prioritization approaches to identify priority areas. Research on biodiversity conservation showed the importance of accounting for costs of land, land use change and the benefits of coordinated identification of priority areas between countries (Strange et al., 2006; Kark et al., 2009; Moilanen et al., 2013; Pouzols et al., 2014). Spatial prioritization approaches have the potential to bridge the science-policy gap (Sinclair et al., 2018). Previous research on spatial prioritization can be broadly divided into research aimed at advancing the scientific field (42%) and research intended for implementation (58%) (Sinclair et al., 2018). The majority (74%) of research intended for implementation actually results in conservation actions (Sinclair et al., 2018). Thus, prioritization approaches can provide important scientific and policy relevant insights to conserve biodiversity and ESs.

Prioritization approaches were originally developed for biodiversity conservation, but have also been applied to ESs. Much of this research has focused on how to integrate ESs alongside biodiversity (Chan et al., 2011; Cimon-Morin et al., 2013; Manhães et al., 2016). Other studies have identified priority areas solely for ESs, without considering biodiversity (Chan et al., 2006; Casalegno et al., 2014; Cimon-Morin et al., 2014, 2016). However, prioritization for ESs differs in important ways from prioritization for biodiversity.

A first important difference is the need to account for ES demand in the identification of priority areas. Land or land systems are multifaceted and are shaped by the interactions between biophysical and socio-economic conditions (Verburg et al., 2013, 2016; Seppelt et al., 2016). Therefore, identifying priority areas and actions for land management need to consider both biophysical and socio-economic conditions. ESs are a combination of the capacity of the land to provide a service and the societal demand for a service.

Identifying priority areas for ESs is complicated by the fact that areas of ES demand and ES capacity are not always in the same locations. The spatial connection between areas with ES demand and ES capacity is referred to as ES flow. Previous studies have primarily used ES capacity to identify priority areas without considering ES demand (Egoh et al., 2010; Adame et al., 2014; Casalegno et al., 2014), whereas others have used ES flow (Cimon-Morin et al., 2014; Schröter et al., 2014b). In Canada, researchers found large differences between identifying priorities based on ES capacity or ES flow (Cimon-Morin et al., 2014). Therefore, it is important to explicitly consider ES flow when prioritizing areas for ESs.

Accounting for ES demand in prioritization studies also means accounting for the distribution of priority areas across locations. Benefits from ESs with local to regional flows are spatially restricted. Depending on the ES considered flows can range from local to global and address the need of different beneficiaries (Serna-Chavez et al., 2014). Maintaining flood regulation within a watershed in the study area does not maintain flood regulation across all watersheds in the study area. In other words, flood protection in The Netherlands does not necessarily benefit people living along the banks of the Ebro. Therefore, prioritization approaches need not only account for ES demand but also for the spatial extent of a particular ES flow. This indicates that prioritizing areas for ESs needs to balance prioritization between multiple regions. Specifically, prioritization approaches need to account for how to distribute benefits and priority areas across beneficiaries and identify winners and losers (Vollmer et al., 2016; Benton et al., 2018). This has not been done in previous studies identifying priority areas for ESs. Therefore, to what extent considering ES flow and the spatial restriction thereof affects the identification and effectiveness of priority areas remains unclear to date.

A second important difference is the impact of land use change on ESs. Land use change is a strong driver of changes in ESs over time (Polce et al., 2016). Impacts of land use change on ESs have been shown to vary strongly depending on the region and the ESs studied (Stürck et al., 2015b; Polce et al., 2016). Following land use change, relations between ESs can shift from trade-offs to synergistic relationships and vice versa (Renard et al., 2015). An assessment of threats through land use change is however crucial for coupling the identification of priority areas with actions to potentially conserve these areas (Luck et al., 2012).

Identification of priorities is mostly based on a static assessment of ESs under current land use (Chan et al., 2006; Casalegno et al., 2014; Schröter et al., 2014b). Only very few prioritization studies for ESs consider land use change (Cimon-Morin et al., 2016; Fan et al., 2016). Studies on prioritization for biodiversity conservation have accounted for threats of land use change (Faleiro et al., 2013; Pouzols et al., 2014; Zhu et al., 2015). In general, these studies assign uniform negative weights to areas with land use change, irrespective of the species and the type of land use change considered. However, assigning uniform negative weights to areas with projected land use change cannot be applied to ESs, because it would ignore the fact that each ESs responds differently to land use change. It is important to consider the actual impacts of land use changes on ES capacity and flow. This includes the anticipation that these may be different depending on the ES and the location studied. Moreover, studies on land use change and prioritization focus primarily on changes in land cover alone, ignoring the effects of changes in configuration and management intensity. This hampers our understanding of the level and the origin of changes in ESs over time and the consequences thereof for prioritization. Currently, insights in the effects of land use change on prioritizing areas for ES are unknown and are important next steps in operationalizing these methods for ES policy and practice.

1.3.2 Prioritizing areas and actions for ecosystem services

ES maps are also used as an input to identify priorities for restoring landscapes. As suggested by the name, methods on spatial conservation prioritization mostly focus on conservation areas and actions. However, landscapes can also be restored to maintain ESs over time. For restoration the focus is often on prioritizing areas (where to restore) and actions (how to restore) simultaneously. Different methods are available to prioritize areas and actions together. Here I distinguish between scenario analysis and optimization approaches.

Scenario analyses are applied to compare outcomes of restoration and conservation actions (Lautenbach et al., 2013; Zhu et al., 2015; Barnett et al., 2016; Tekalign et al., 2017). Scenario analysis are used either to compare alternative conservation actions or to assess the effect of external driving factors on conservation outcomes. Scenario analysis have been applied in the context of conserving and restoring land for ESs, such as forest restoration in the US or agroforestry expansion in Ethiopia (Barnett et al., 2016; Tekalign et al., 2017). Scenario studies are also used to directly inform policies on conservation and restoration of biodiversity and ESs. For example, scenarios were used to highlight the need for additional expansion of green infrastructure in Europe to maintain ES over time (Maes et al., 2015a), or to assess the degree to which certain policy options can support no-net-loss objectives for ESs

(Schulp et al., 2016). An issue with scenario approaches is that they can only identify a limited set of prescribed possible futures. This limited set applies to either where the action is taken and what action is applied (Seppelt et al., 2013). Therefore, scenarios can only provide insights for a set of prescribed alternatives.

An alternative is to combine scenario analysis with landscape optimization approaches. Landscape optimization algorithms have the potential to compare and test the full array of restoration and conservation alternatives (Seppelt et al., 2013). Landscape optimization can provide decision makers with insights on the best possible way to adapt land management for ESs, both in terms of what action to take and where to take this action. A combination of scenario analysis and landscape optimization analysis can provide information on the difference between existing restoration scenarios and what could potentially be achieved (Seppelt et al., 2013). If the scenarios can be further improved this means that either for the same area restored a higher level of ESs can be achieved. Alternatively, the same amount of ESs can be achieved with a smaller area restored.

Landscape optimization approaches have mostly been applied to study trade-offs between agricultural production and other ESs. These approaches are specifically designed to optimize the landscape for multiple objectives (here: ESs) and to minimize trade-offs between competing objectives Therefore, use of these algorithms for landscape management and restoration is especially interesting in the context of multifunctional intensively used landscapes, where actions need to balance between multiple, often competing objectives (Cord et al., 2017). Landscape optimization has been used to study alternatives for bioenergy expansion or to increase ESs and biodiversity in agricultural landscapes (Polasky et al., 2008; Lautenbach et al., 2013; Pennington et al., 2017). Previous research highlighted the benefits of optimizing landscapes for multiple objectives simultaneously and the potential of this approach to minimize trade-offs (Polasky et al., 2008; Kennedy et al., 2016; Law et al., 2017; Pennington et al., 2017). In Brazil, trade-offs between agricultural production, biodiversity and freshwater ESs could be largely overcome, but only when all these objectives where jointly included in the landscape optimization (Kennedy et al., 2016). Landscape optimization approaches can thus explore the potential of designing sustainable (production) landscape through explicitly changing land use and land management patterns (Turner et al., 2013; Pennington et al., 2017).

Studies on landscape optimization have mostly looked at changes in the allocation of land cover. To what extent changes in land cover, landscape configuration and management intensity together can contribute to restoring ESs in intensively used landscapes is unclear to date. Moreover, none of the studies have looked at the potential of small-scale landscape elements in restoring ESs in these landscapes (Jones et al., 2013). Therefore, integrating landscape configuration and management intensity into landscape optimization approaches can provide important insights into the opportunities to restore and maintain multifunctional production landscapes.

1.3.3 Using ES maps to identify priority areas and actions

To summarize, ESs have increasingly been included in spatial optimization methods aimed at where and how to conserve or restore the land. These spatial optimization approaches encompass methods related to systematic conservation planning (SCP) and landscape optimization algorithms. A main benefit of these approaches is that they are specifically designed to integrate and balance between multiple objectives (here: ESs).

Challenges remain in how to integrate ESs into these approaches. Currently we lack an understanding of how priority areas based on ES flow and ES capacity differ. Moreover, we need to develop novel methods such that SCP approaches can further integrate ES flow and the need to distribute conservation areas and actions across beneficiaries. An alternative to identify priority actions for restoration is to use landscape optimization algorithms. A main advantage of both SCP and landscape optimization is that these methods are specifically designed to balance multiple objectives and minimize trade-offs. So far these approaches have mostly looked at linkages between land cover and ESs. Integration of landscape configuration and management intensity is limited. This hampers our understanding of potential threats and restoration actions to maintain ESs and limits the support for decision makers on how to manage our land for multiple ESs.

Improving methods to identify ESs priorities serves a scientific purpose. In addition, it also aims to improve decision making on these matters. However, better tools and better scientific understanding do not necessarily result in better decision making. Prioritization approaches have the potential to bridge the science-policy gap, given that much of the research on systematic conservation planning is used in actual decision making on land management (Sinclair et al., 2018). But the use and uptake of any spatial decision support tool for land management depends on wider considerations such as the ease of use and the political process and timing (Primmer and Furman, 2012; Rosenthal et al., 2015; Bouwma et al., 2018). Irrespective of the uptake in decision making, findings from prioritization approaches that integrate ESs can make trade-offs between ESs more explicit and create a broader view on the pros and cons of different land management alternatives.

1.4 Objective of the thesis

The overall objective of this thesis is to develop and apply methods for integrating multiple ESs into landscape prioritization and optimization methods, with the aim to identify effective locations and management actions for the conservation and restoration of ecosystem services. Specifically, the thesis focuses on the role of land use change, landscape configuration and management intensity in a context of societal demands for these services. Throughout this thesis I focus on multiple ESs. I study how all facets of land can be incorporated in optimization and prioritization methods and how the identified priorities play out for multiple ESs simultaneously. To meet this goal, I address the following research questions:

(RQ1) How does the spatial configuration of land cover affect the capacity of the landscape to provide ESs, and what are the implications thereof for mapping ESs?

(RQ2) How are the trade-offs between ESs affected by land management intensity?

(RQ3) How do societal demands for ESs, landscape configuration and management intensity change the relative priority of areas for ESs?

(RQ4) What are the opportunities in multifunctional landscapes for restoration through changes in land cover, landscape configuration and management intensity?

Below I explain how each chapter contributes to the questions posed (see also Figure 1.1).

1.4.1 Structure of the thesis

This thesis can be broadly divided into two main sections. The first two chapters aim to improve the assessment of ESs. Chapter 2 studies the importance of landscape configuration on ES capacity and how accounting for landscape configuration affects mapping ESs. First, we review the evidence base for an effect of landscape configuration on individual ESs. Next, those ESs affected by landscape configuration are mapped with and without accounting for landscape configuration. Chapter 2 contributes directly to research question 1 by reviewing and quantifying the effects of landscape configuration on ESs. Findings from this chapter are important input to questions 3 and 4.

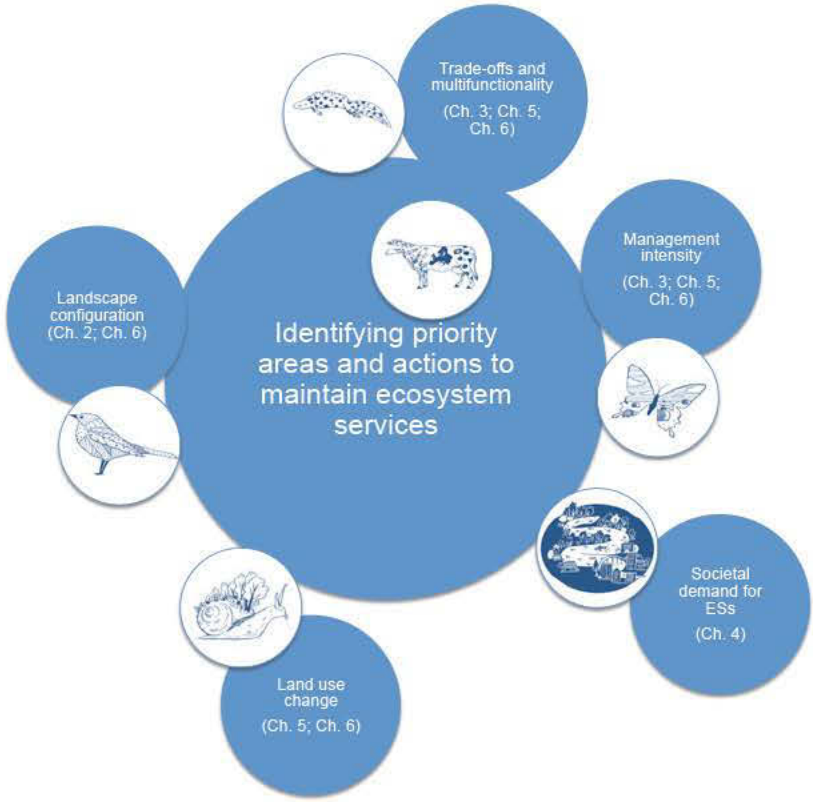
Chapter 3 presents a meta-analysis on the trade-off between biodiversity and food, feed and fibre production. Specifically, we study how this trade-off is mediated along a gradient of management intensity. We quantify the response of biodiversity (species richness) and production across individual case studies under at least two levels of management intensity. These responses were studied in crop production, grass production and forest production systems. Here, we focussed on provisioning services and biodiversity, given that many services depend on biodiversity (Cardinale et al., 2012). Moreover, previous studies highlighted that land use intensification can have affect ESs through changes in biodiversity (Allan et al., 2015; Chillo et al., 2018). The findings of this chapter directly contribute to research question 2 and provide a first estimate of effects of land management on biodiversity and ESs.

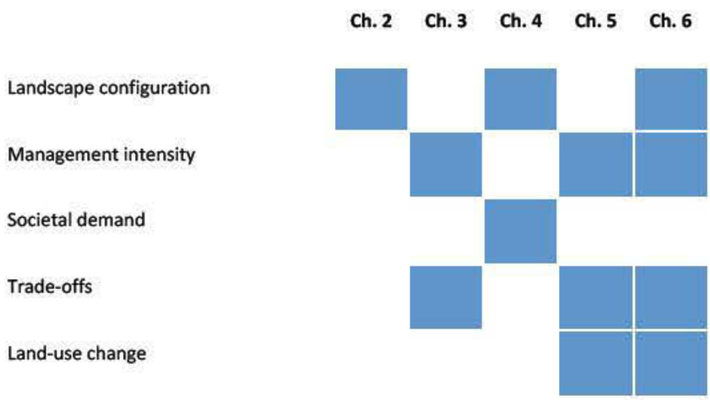
The second part of the thesis studies how to identify priority areas and actions to manage land for multiple ESs. We study how societal demand, landscape configuration and management intensity affect the identification of priority areas and actions. In Chapter 4 the focus is on identifying priority areas for ES conservation, and the importance of accounting for societal demand and ES flow in doing so. Here, existing systematic conservation planning (SCP) methods are used for the first time to identify priority areas for ESs at the scale of the European Union. In chapter 4 we compare alternative approaches to integrate the configuration of ES demand and ES capacity in SCP approaches and introduce a method to account for distribution of priority areas across ES flow zones. Findings from this chapter directly contribute to research questions 1 and 3 by providing additional evidence for effects of configuration on ES capacity and ES flow.

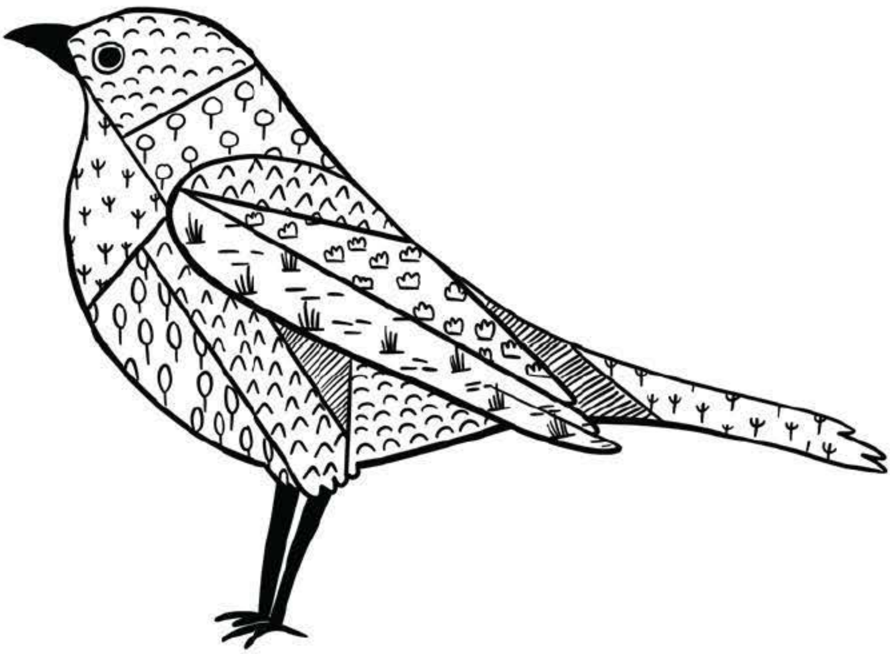
In Chapter 5 we quantify the effects of land use change on the identification of priority areas for ESs. Specifically we study how locations of priority areas and the ESs maintained therein change over time. We assess the separate effects of changes in management, land cover and landscape configuration. Findings from this chapter provide a direct answer to research question 3. In contrast to other chapters this study directly accounts for scenarios of land use change. It identifies areas suitable for conservation and areas suitable for restoration.

In Chapter 6 we shift the focus towards restoring ESs in agricultural landscapes. Chapter 6 concerns the challenge of how to optimally allocate land management options in an agricultural landscape for multiple competing objectives. Here we compare management actions focused on allocating land use change, management intensity and landscape configuration for their potential to restore multiple ESs. This study supported the actual implementation of land management options in three ways: (i) through the provision of quantitative estimates of the effects of measures on multiple objectives (ii) by providing an indication of the biophysical limits to multifunctionality; and (iii) by exploring whether the current nature management plan could be strengthened. Lastly, the study presents a novel method to identify spatial priorities for restoration based on the wide set of optimal landscape configurations. Findings from this chapter provide direct input into research question 4.

Figure 1.1: Framework of the thesis. Central is the overall topic of this thesis. The individual points depict the main topics addressed in this thesis, and the respective chapters (Ch.) in which this topic is addressed. The table provides a quick overview of which topics are addressed in which chapters of the thesis.







2. Landscape configuration and ecosystem services

Abstract

Humans structure landscapes for the production of food, fibre and fuel, commonly resulting in declines of non-provisioning ecosystem services (ESs). Heterogeneous landscapes are capable of providing multiple ESs, and landscape configuration—spatial arrangement of land cover in the landscape—is expected to affect ES capacity. However, the majority of ES mapping studies have not accounted for landscape configuration. Our objective is to assess and quantify the relevance of configuration for mapping ES capacity. A review of empirical evidence for configuration effects on the capacity of ten ESs reveals that for four ESs configuration is relevant but typically ignored in ES quantification. For four ESs we quantify the relevance of configuration for mapping ESs using Scotland as a case study. Each ES was quantified through modelling, respectively ignoring or accounting for configuration. The difference in ES capacity between the two ES models was determined at multiple spatial scales. Configuration affected the capacity of all four ESs mapped, particularly at the cell and watershed scale. At the scale of Scotland most local effects averaged out. Flood control and sediment retention responded strongest to configuration. ESs were affected by different aspects of configuration, thus requiring specific methods for mapping each ES. Accounting for configuration is important for the assessment of certain ESs at the cell and watershed scale. Incorporating configuration in landscape management provides opportunities for spatial optimization of ES capacity, but the diverging response of ESs to configuration suggests that accounting for configuration involves trade-offs between ESs.

This chapter is published as:
Verhagen, W., Van Teeffelen, A. J. A., Baggio Compagnucci, A., Poggio, L., Gimona, A., & Verburg, P. H. (2016). Effects of landscape configuration on mapping ecosystem service capacity: a review of evidence and a case study in Scotland. *Landscape Ecology*, *31*(7), 1457-1479. DOI: 10.1007/s10980-016-0345-2

2.1 Introduction

Many regions in the world have been transformed to large, homogeneous agricultural areas to meet increasing demands for food, fibre and fuel (Klein Goldewijk, 2001; Foley et al., 2005; Monfreda et al., 2008). The focus on providing food, fibre and fuel has often resulted in declines of non-provisioning ecosystem services (ESs) (Foley et al., 2005; Rodríguez et al., 2006; Bennett et al., 2009; Power, 2010). In the upcoming decades pressures on ES capacity, the potential of an ecosystem to supply an ecosystem function or service without consideration of ES demand, are expected to grow, while demand for almost all ESs is increasing (Millenium Ecosystem Assessment 2005; Alcamo et al. 2005). Some landscapes provide high levels of a single ES. Interest in multifunctional landscapes, capable of providing multiple ESs simultaneously, is rising (Bennett et al., 2009; Jones et al., 2013; Qiu and Turner, 2013; Schindler et al., 2014). A multifunctional landscape is thought to be affected by its spatial heterogeneity and can be managed through the landscape structure (Macfadyen et al., 2012; Jones et al., 2013). Unravelling the relation between ESs and landscape heterogeneity is crucial for determining the promises of multifunctional landscapes (Bennett et al., 2009; Jones et al., 2013).

Heterogeneity in land cover within and between landscapes can affect ES capacity both directly and indirectly. Heterogeneity directly affects ES capacity through ecological processes (e.g. the flow and retention of water and nutrients) or indirectly through positive effects on biodiversity (Wiens, 2002; Lovett et al., 2005; Fahrig et al., 2011; Stein et al., 2014). Species richness is positively correlated to heterogeneity in land cover (Stein et al., 2014) and, although many uncertainties remain, species richness and other biodiversity indicators are positively correlated to ESs (Balvanera et al., 2006, 2014; Cardinale et al., 2012; Mace et al., 2012; Harrison et al., 2014). These direct and indirect effects suggest that the capacity of some ESs is affected by landscape heterogeneity. Landscape heterogeneity can be broken down into two components: 1) the composition, in terms of the type(s) of land cover; and 2) the configuration, the spatial arrangement of land cover types (Fahrig and Nuttle, 2005; Jones et al., 2013).

The mapping of ESs is a common tool to assess ES capacity. Although many studies focus on mapping the capacity of ESs, these studies commonly do not account for landscape configuration. Mapping ES capacity is often linked to the landscape composition, using land cover proxies (Seppelt et al., 2011a; Martínez-Harms and Balvanera, 2012). The link between ES capacity and landscape configuration has been studied in field experiments (Liu et al., 2012; Andersson et al., 2013; Mitchell et al., 2014b) but is rarely incorporated in studies mapping ESs. In studies mapping ES, the combined effects of landscape composition and configuration were incorporated by relating landscape metrics to ES values (Sherrouse et al., 2011), ecological functioning (Frank et al., 2012) and ES capacity (Laterra et al., 2012). Lautenbach et al. (2011) mapped three ESs using indicators sensitive to landscape configuration. Changes in ES capacity, following land cover change, could not be fully explained by changes in landscape composition alone, concluding that some ESs can only be quantified if indicators account for configuration as well (Lautenbach et al., 2011). Recently, Mitchell et al. (2015a) showed that fragmentation affects ESs capacity at the landscape level and at the cell level by modelling the capacity of an ES for a set of hypothetical landscapes. So far a comprehensive assessment of the relative effects of composition and configuration on a wider set of ESs is missing. The absence of studies on the relation between the components of landscape heterogeneity and ES capacity hampers landscape management for multiple ESs and fails to provide specific guidance beyond generic ideas like “heterogeneity is good” (Macfadyen et al., 2012).

The aim of this paper is to assess and quantify the importance of landscape configuration for mapping the capacity of ESs. In particular, we aim to distinguish between the effects of landscape composition and configuration on ESs in approaches to map ESs across larger regions. We start with a short overview of empirical evidence relating landscape configuration to ES capacity, focusing on studies in temperate and continental climates. Based on the evidence from field studies we account for the effect of landscape configuration on mapping ES capacity by comparing two ES models, one with and one without landscape configuration. Scotland is used as case study site.

2.2 Empirical evidence for a relation between landscape configuration and ES capacity

2.2.1 Review approach

To determine which ESs can be expected to be sensitive to landscape configuration, we performed a literature search for field studies and reviews relating landscape configuration to ES capacity. For ten commonly studied ESs with relevance to Scotland (Aspinall et al., 2011; Seppelt et al., 2011a), we searched the ISI Web of Science database (timeframe: 1990 – May 2014), using a variety of search terms for landscape configuration, accounting for the diversity in terminology in the literature (see S1). For selection and reviewing of papers, we primarily focused on the relation between landscape configuration and ES capacity, but, if relevant, we also incorporated effects of landscape heterogeneity or compositional heterogeneity. Of the first 50 papers returned per combination of ES and landscape configuration term (as ranked by 'relevance' in Web of Science) we scanned title and abstract. Since we use Scotland as a case study area we focused our review on studies from temperate and continental climates. When studies from the tropics were included this is mentioned in the text. Following previous literature reviews (Harrison et al., 2014; Stein et al., 2014), the set of papers was next expanded using the snowballing technique The snowballing technique was used because the initial set of papers primarily returned papers on general search terms, whereas papers on specific terms, such as patch size, were largely absent.

Based on the literature review we distinguished four aspects of configuration that can affect ES capacity. First, ES capacity is affected by the specific location of land cover types (e.g., Ricketts et al. 2008; Acreman and Holden 2013). Examples include discrete classes (riparian vs. non-riparian) or distance between land cover and a specific feature (e.g. roads, farms or streams). Second, ES capacity is affected by the use and structure of multiple patches (e.g., Bodin et al. 2006; Ludwig et al. 2007; Liu et al. 2012; Bateni et al. 2013), such as foraging behaviour or nutrient flows through a landscape. Third, ES capacity is affected by the structure of a single patch, such as patch size or edge effects (e.g., Forman 1995; Macfadyen and Muller 2013). Fourth, ES capacity is affected by the presence of linear elements, such as hedgerows and grass margins (e.g., Falloon et al. 2004; Pollard and Holland 2006; Borin et al. 2010). We included linear elements, because of their effect on the spatial patterning of land cover and their expected importance for a subset of ESs. Below we summarize the main results from the literature review.

2.2.2 Review results

There is evidence for an effect of configuration on ES capacity for four ESs, namely nutrient retention, pollination, landscape aesthetics and sediment retention (Table 2.1). The evidence for an effect of configuration on crop production, flood control and pest control is mixed. No evidence is found for a relation between landscape configuration and carbon sequestration. Our literature search did not yield studies on the relation between landscape configuration and wood production or cattle farming. We first discuss the ESs for which there is no or mixed evidence, followed by ESs affected by landscape configuration.

Table 2.1: Overview of empirical evidence for the dependency of ES capacity on configuration. When evidence exists for a relation between ES capacity and configuration, the type of configuration effect(s) are specified (✓). Individual letters correspond to the key references that substantiate the individual claims per ES (see footnote). For cattle and wood production the search term did not return any papers on the relation to landscape heterogeneity.

	No papers/ No evidence for effect	Mixed Evidence	Evidence exists for effect of			
			Location of land cover	*Multiple patches*	*Single patch*	*Landscape element*
Carbon sequestration[a]	✓					
Cattle production	No papers					
Crop production[b]		✓				
Flood Control[c]		✓	✓			✓
Nutrient retention[d]			✓	✓		✓
Pest Control[e]		✓				
Pollination[f]			✓	✓	✓	✓
Landscape aesthetics[g]				✓		
Sediment retention[h]			✓	✓		✓
Wood production	No papers					

[a](Falloon et al., 2004; Follain et al., 2007; Borin et al., 2010; Laurance et al., 2011; Paula et al., 2011; Lenka et al., 2012; Williams and Hedlund, 2013; D'Acunto et al., 2014; Ziter et al., 2014)

[b](Sparkes et al., 1998; Foereid et al., 2002; Roschewitz et al., 2005b; Borin et al., 2010; Persson et al., 2010; Poveda et al., 2012; Abson et al., 2013; Lemessa et al., 2013; Deguines et al., 2014)

[c](Carroll et al., 2004; Calder, 2007; Marshall et al., 2009; Borin et al., 2010; Mayor et al., 2011; Sriwongsitanon and Taesombat, 2011; Lenka et al., 2012; Acreman and Holden, 2013; von Freyberg et al., 2014)

[d](Castelle et al., 1994; Johnson et al., 1997; Heathwaite et al., 1998; Braskerud, 2002; Uuemaa et al., 2005; Wang et al., 2005; Gergel, 2005; Lee et al., 2009; Borin et al., 2010; Liu et al., 2012; Bateni et al., 2013; Sun et al., 2013)

[e](Thies and Tscharntke, 1999; Roschewitz et al., 2005a; Bianchi et al., 2006, 2010; Pollard and Holland, 2006; Bianchi et al., 2013; Tscharntke et al., 2007; Vollhardt et al., 2008; Perović et al., 2010; Chaplin-Kramer et al., 2011; Poveda et al., 2012; Rusch et al., 2012, 2013; Chaplin-Kramer and Kremen, 2012; Pisani et al., 2013; Veres et al., 2013; Macfadyen and Muller, 2013; Martin et al., 2013; Mitchell et al., 2014b; Morandin et al., 2014)

[f](Svensson et al., 2000; Kells et al., 2001; Potts et al., 2003; Kremen et al., 2004; Bodin et al., 2006; Williams and Kremen, 2007; Ricketts et al., 2008; Winfree et al., 2009; Isaacs and Kirk, 2010; Kennedy et al., 2013; Morandin and Kremen, 2013; Rollin et al., 2013; Bailey et al., 2014; Stanley and Stout, 2014)

[g](de la Fuente de Val et al., 2006; Dramstad et al., 2006; Borin et al., 2010; Kienast et al., 2012; van Zanten et al., 2014)

[h](Castelle et al., 1994; Bartley et al., 2006; Ludwig et al., 2007; Bu et al., 2008; Lenka et al., 2012; Yang et al., 2012; Shi et al., 2013)

No evidence is found for a relation between landscape configuration and carbon sequestration. Soil organic carbon shows no difference along a gradient of landscape heterogeneity (Williams and Hedlund, 2013) and hedgerows only locally increase soil organic carbon (D'Acunto et al., 2014). In tropical systems forest fragmentation and edge effects result in a decrease in carbon sequestration (Laurance et al., 2011; Paula et al., 2011) but preliminary evidence suggests that in temperate regions carbon sequestration is unaffected in forest edges (Ziter et al., 2014).

Crop production is affected by configuration but these effects are often indirect, due to time lags or ES interactions. In the UK, returns from crop production are reduced, but show less annual variation in landscapes with a higher diversity of agricultural land uses (Abson et al., 2013). Increases in crop production in France - following the disappearance of semi-natural habitat and linear elements - decline with increasing dependency of the crop on pollination (Deguines et al., 2014). Crop production is reduced at field edges, especially when the field edge is adjacent to tree lines or hedges (Sparkes et al., 1998; Foereid et al., 2002), but in general effects are considered to be local and small compared to total crop production (Borin et al., 2010).

Pest control constitutes of the interaction between pest species and natural enemies. A review and meta-analysis of pest control studies both show that natural enemy populations respond positively to increasing landscape complexity (Bianchi et al., 2006; Chaplin-Kramer et al., 2011). An increase in natural enemy populations does not necessarily translate into increased pest control, because pest species populations can also respond positively to increasing landscape complexity (Bianchi et al., 2006; Chaplin-Kramer et al., 2011). The same issue applies to studies on linear elements and pest control, although Morandin et al. (2014) show that fields adjacent to hedgerows less frequently reach pest pressure levels that require insecticides use.

For flood control there is evidence for an effect of configuration on ES capacity. The location of land cover affects runoff. For example, the percentage of rainfall that resulted in runoff decreased with increasing upstream area, attributed to infiltration of runoff along the flow path (Mayor et al., 2011). Linear elements can greatly reduce runoff (Borin et al., 2010) and in the UK the presence of individual trees and shelterbelts increases the infiltration capacity of grazed pastures (Marshall et al., 2009). The effect of land cover depends however on the amount of rainfall and diminishes with increasing soil saturation (Lull and Reinhart, 1972; Calder, 2007; Acreman and Holden, 2013). In mountainous catchments the riparian zone dominates the runoff response and intercepts high amounts of nutrients but also contributes most to runoff during conditions of high soil saturation (von Freyberg et al., 2014). Forests can store large amounts of water but during larger rainfall events forest areas in the tropics produce more runoff than non-forested areas because of quick soil saturation and increased base flow (Sriwongsitanon and Taesombat, 2011).

There is evidence for a relation between landscape configuration and the ESs nutrient retention, sediment retention, pollination and landscape aesthetics. Each ES is however affected by different aspects of landscape configuration (Table 2.1). Sediment retention and nutrient retention, here limited to nitrogen, respond largely similar to configuration. Retention services are affected by the location of land cover, especially riparian vegetation (Castelle et al., 1994; Johnson et al., 1997) and the structure of multiple patches, both at landscape scale (Liu et al., 2012; Bateni et al., 2013) and at the hillslope scale in the tropics (Bartley et al., 2006; Ludwig et al., 2007). Buffer strips and wetlands can intercept high amounts of nutrients and sediment (Castelle et al., 1994; Heathwaite et al., 1998).

Pollination, here limited to pollination by wild bees, is affected by all aspects of landscape configuration. Pollination capacity is affected by the location of land cover types. Abundance and visitation rates of bees to cropland strongly decrease with increasing distance between cropland and bee habitat (Ricketts et al., 2008). Moreover, pollination capacity is affected by the structure of multiple and single patches. Bees require both nesting sites and floral resources in close proximity (Potts et al., 2003; Williams and Kremen, 2007). Forest edges typically harbour more nesting opportunities and floral resources than interior forest sites (Svensson et al., 2000; Kells and Goulson, 2003).

Last, linear elements can provide important habitat for bee species and reduce flight distances. Stanley and Stout (2014) found large overlap between pollinators visiting oil seed rape crops and wild flowers in hedgerows, even during mass flowering of the crops, suggesting that hedgerows can be an important additional floral resource.

Landscape aesthetics, here a combination of recreation potential and landscape aesthetics, is influenced by the structure of multiple patches. Several studies stress the importance of compositional and configurational heterogeneity for recreation (de la Fuente de Val et al., 2006; Dramstad et al., 2006; Kienast et al., 2012).Moreover, a meta-analysis on landscape preferences showed that a mosaic landscape is more appreciated than either an agricultural or natural dominated landscape (van Zanten et al., 2014). The presence of linear elements resulted in a wide variety of responses in landscape preferences, making generalization difficult (van Zanten et al., 2014).

Table 2.2: Number of ecosystem service mapping studies that account for configuration in the indicators used, and the type of configuration effect that is accounted for. The total number of studies that account for configuration per ecosystem service does not have to equal the type of configuration effects incorporated, because a single study can account for multiple configuration effects. No effect indicates that the ES mapping study does not account for configuration.

	# of studies	No effect	Accounts for effect of			
			Location of land cover	*Multiple patches*	*Single patch*	*Landscape elements*
Flood Control	5	4	1			0
Nutrient Retention	6	2	4	3		0
Pollination	3	2	1	1	0	0
Landscape Aesthetics	6	6		0		
Sediment Retention	11	8	3	2		0

2.2.3 Comparison to ES mapping studies

We performed a review of ES mapping studies to determine whether these studies account for configuration. Studies mapping ES capacity were selected from a database of 271 ES case studies published until 1.8.2013 (Seppelt et al., 2011a; Lautenbach et al., 2019). Of the 271 papers in the database, 73 studies mapped ES. We incorporated studies mapping nutrient and sediment retention, landscape aesthetics and pollination. Furthermore, we included studies mapping flood control because configuration affects flood control under certain conditions. Per mapping study we checked whether the study accounted for configuration in mapping the ESs and which of the four aspects of configuration were incorporated. The majority of ES mapping studies (65 %) does not account for configuration in mapping ES capacity (Table 2.2). Studies that account for configuration do not always account for all aspects of configuration. None of the studies mapping landscape aesthetics capacity account for configuration. Only for nutrient retention the majority of studies (73%) accounts for configuration. For nutrient and sediment retention, studies that account for configuration commonly use the InVEST model (Kareiva et al., 2011). Linear elements are never incorporated in the assessment. Based on the findings from the two literature reviews we conclude that there is a potential gap in accounting for configuration for mapping the ES capacity of flood control, sediment retention, landscape aesthetics and pollination.

2.3 Quantifying the effect of configuration on ES mapping - Methods

2.3.1 Mapping ES capacity

Following the results of the literature review we quantified and mapped the capacity of four ESs for Scotland: flood control, sediment retention, landscape aesthetics and pollination. Nutrient retention was omitted because the configuration effect. Land cover was obtained from the 2007 UK land cover map (lcm2007) raster version at 25 meter resolution (referred to as "cell" hereafter) (CEH, 2011).

The composition models for all ESs were largely based on land cover proxies: each cell was assigned a single value per ES, based on its land cover. Land cover proxies do not account for configuration, i.e. the projected ES capacity is always the same for a cell of a given land cover type, irrespective of landscape configuration (Burkhard et al., 2009). In other words, the ES land cover proxies balance all possible landscape configurations and ultimately represent the on-average effect of landscape configuration. In all models, we assumed that the composition model represents the ES capacity based on the landscape composition and the average configuration effect. Hence, the larger the deviations of the configuration model from the composition model, the larger the effect of accounting for configuration on ES capacity. Following this rationale, accounting for landscape configuration can increase or decrease a cell's ES capacity, as projected by the composition model. We refer to an increase (decrease) as a positive (negative) effect of landscape configuration on ES capacity. Per model we explain the calculation of the average configuration effect below. Detailed descriptions of all composition and configuration models are provided in S2.

Sediment retention capacity - Sediment retention capacity was mapped using InVEST (Kareiva et al., 2011). The InVEST model has been extensively documented (Kareiva et al., 2011), but a short summary of the main components is included here. In the InVEST model, sediment retention consists of (i) sediment retention at the cell and (ii) filtration of sediment input from upstream cells. Sediment retention at the cell was calculated using the revised universal soil loss equation (Renard et al., 1997), based on rainfall, soil erodibility, topography, the land cover and the land management. Sediment retention at the cell is calculated as the difference in soil loss of a cell with and without accounting for the effect of the cell's land cover. Sediment filtration was calculated based on a land cover proxy and the sediment input from upstream cells. The land cover proxy for filtration was assigned to each land cover type based on a combination of literature (May and Place, 2005; Anderson et al., 2010; Sude et al., 2011), documentation of the InVEST model and an expert assessment. Sediment input depends on the position of the cell in the landscape and the land cover of surrounding cells. Per watershed in Scotland we calculated the average sediment input of all cells in the watershed. In the composition model, all cells within a watershed were assigned this average sediment input value. In the configuration model, sediment filtration per cell was calculated using the actual sediment input per cell.

Flood control capacity - In the composition model, flood control capacity was quantified using land cover proxies for flood protection from Burkhard et al. (2012). Burkhard et al. (2012) assigned a value (0-5) per ES to each CORINE land cover type, based on field studies and expert assessment. The values from the ES matrix were divided by five to range from 0-1, as has been done by others before (Schulp et al., 2014a; Stoll et al., 2014). As we used lcm2007 land cover data, we transferred the land cover proxies per CORINE land cover class to the closest lcm2007 class (S1). In the configuration model, flood control capacity of a cell depended on the land cover of that cell (composition model) and on the location of the cell along the flow path. Following Chan et al. (2006), we calculated the flow accumulation value (FACC) per cell as the number of upstream cells, irrespective of land cover type, potentially transporting water into a single cell. The FACC accounts for the position of the cell within a watershed relative to the flow path. Per watershed we calculated the average FACC value of all cells in the watershed. The FACC value of a cell was scaled relative to the average FACC of the watershed to which the cell belonged, indicating that cells with a high amount of upstream area contribute more to flood control. Modelled effect of landscape composition and configuration do not apply under conditions of full soil saturation.

Landscape aesthetic capacity - In the composition model landscape aesthetic capacity was quantified using land cover proxies for "landscape aesthetics and inspiration" from Burkhard et al. (2014). As for flood control, we transferred the land cover proxies per CORINE land cover to the closest lcm2007 land cover. In the configuration model, landscape aesthetics capacity depended on the land cover proxy (composition model) and the surrounding land cover diversity. Land cover diversity assigned to a cell was determined by two factors: the landscape type surrounding a cell, and the land cover diversity within that landscape type. First, we assigned each cell to one of three generic landscape types: natural dominated landscape, agricultural dominated landscape or mosaic landscape. The landscape type was calculated within a view shed of 200 meters following Casado-Arzuaga et al. (2013). Agricultural dominated landscapes had > 50% agricultural land cover in the view shed, whereas natural dominated landscapes had >50% natural land cover in the view shed. In mosaic landscapes none of the two land covers, agriculture or natural, dominated. We obtained landscape preference scores per landscape type from a meta-analysis for European agricultural landscapes (van Zanten et al., 2014). The landscape preferences were in the order mosaic (most preferred), natural, agricultural (least preferred) (van Zanten et al., 2014). Each cell was assigned a landscape preference score based on the landscape type it belonged to. Second, we calculated the Shannon diversity index (SHDI) per cell based on land cover, again within a view shed of 200 meters. Third, per cell, we calculated the overall land cover diversity score by normalizing (min-max normalization) the SHDI value using the landscape preferences for the generic landscape type of that cell (minimum) and mosaic landscapes (maximum). Fourth and last, the overall land cover diversity score per cell was scaled relative to the average land cover diversity score for Scotland. The landscape preference scores from the meta-analysis by van Zanten et al. (2014) apply to European agricultural landscapes. Due to a lack of other data sources on the effect of land cover diversity on landscape aesthetics potential, we applied the same methodology to all landscapes in Scotland.

Pollination capacity - Pollination capacity per cell was quantified with land cover proxies from Zulian et al. (2013), who present separate land cover proxies for nesting suitability and floral resource availability for bees per CORINE land cover type. In the composition model, pollination capacity was mapped using the nesting suitability scores. Following Lautenbach et al. (2011), nesting suitability values for wild bees were not assigned to cropland. As for landscape aesthetics and flood control, we transferred the land cover proxies per CORINE land cover to the closest lcm2007 land cover. In the configuration model, pollination capacity was quantified using the InVEST pollination model (Lonsdorf et al., 2011). In InVEST, pollination capacity of a cell depends on the nesting suitability of a cell, floral resource availability within the surroundings of nesting cells, and the distance between nesting cells and cropland. Land cover proxies for floral resource availability were obtained from Zulian et al. (2013). Compared to the InVEST model, three adjustments were made. First, following Zulian et al (2013) we accounted for edge effects by assigning separate values for nesting suitability and floral resource availability to forest interior and edge cells. Forest edge cells were defined as those cells within 50 meters from other land cover types. Second, in InVEST the effect of floral resource availability on pollination capacity declines with distance (Lonsdorf et al., 2011). We applied a maximum bee flight distance of 500 meters following Lautenbach et al. (2011). Per cell, we additionally calculated the average floral resource availability within a 500-meter radius from the nesting cell without accounting for distance decay or edge effects. We adjusted the InVEST model by scaling the floral resource availability per nesting cell, after accounting for distance decay and edge effects, relative to the average floral resource availability. Third, in InVEST pollination capacity of a site declined with increasing distance from cropland cells (Lonsdorf et al., 2011). In contrast to InVEST, we assigned pollination capacity scores to the nesting cells and not to cropland cells. Furthermore, we calculated the average distance decay effect for distance to cropland, again using a 500-meter radius. If the 500-meter radius would be split at 353.55 meter, the area of the inner and outer circle would be equal and both circles could potentially hold the same amount of cropland cells. The average distance decay effect is therefore calculated as the distance decay effect at 353.55 meter. The effect of distance to cropland per cell was scaled relative to the average distance decay effect. In both the composition and configuration model, pollination capacity was only assigned to nesting sites within 500 meters of cropland.

2.3.2 Comparing ES composition and configuration models

We compared the ES composition and configuration models to assess the effect of accounting for configuration on the level of ES capacity and the spatial pattern of ES capacity across scales. First, we calculated the effect of accounting for configuration on mapping ES capacity per cell as the difference in the ES capacity between the configuration and composition model divided by the ES capacity of the composition model (referred to as "relative effect"). A positive (negative) relative effect of configuration means that accounting for configuration increases (decreases) the ES capacity. Second, we mapped the percentage difference per cell to assess spatial patterns across Scotland and within a single watershed. The percentage difference is simply the relative effect multiplied by 100 percent. Third, we calculated the average of the absolute percentage difference in ES capacity for all cells and watersheds, and for Scotland as a whole. The last analysis account for the absolute effect of configuration across scales and does not account for positive and negative effects of configuration on mapping ES capacity.

Next to the comparison of the ES models we tested whether the percentage difference between the composition and configuration model per cell could be approximated using landscape metrics. We calculated the correlation between three landscape metrics and the difference in ES capacity using Pearson's correlation coefficients. A high correlation would indicate that a particular landscape metric could be used to account for the effect of configuration on mapping that ES. We selected three landscape metrics: a landscape composition metric (% natural vegetation, see S22 section "landscape aesthetics capacity model" for land cover classes assigned to natural vegetation), a compositional heterogeneity metric (land cover richness) and a landscape configuration metric (patch density). Patches were identified as cells of identical land cover within an eight cell neighbourhood. Patch density equalled the total number of patches in an area (summed over all land cover types). According to results from our literature review, the structure of multiple patches, including patch density, was related to sediment retention, pollination and landscape aesthetics. A preliminary analysis showed that patch density is highly correlated with edge density. Previous research showed that edge density is a good predictor of sediment retention (Uuemaa et al., 2005; Liu et al., 2012). Correlations with many more landscape metrics could have been tested, but we decided to only select three metrics that are readily explained, and hence can serve to draft hypotheses on the relation between configuration and ES capacity. For the services flood control and sediment retention we additionally calculated the distance per cell to the closest stream or river per watershed ("distance to water"). The correlations were calculated for a sample of the data to reduce potential bias from spatial autocorrelation. We sampled 10% of all cells with a minimum distance of 100 meter between each cell. For pollination, a separate sample was taken only from the nesting sites within 500 meter of cropland. Per cell, the landscape metrics were calculated on the land cover within 250, 500 and 1000 meter radii to assess the sensitivity of the correlation for the selected radii. All statistical analysis were conducted in R (R Core Team 2013) including the additional packages 'bigmemory' (Kane et al., 2013) and 'reshape'(Wickham, 2007).

2.4 Results

2.4.1 Difference between the ES composition and configuration models

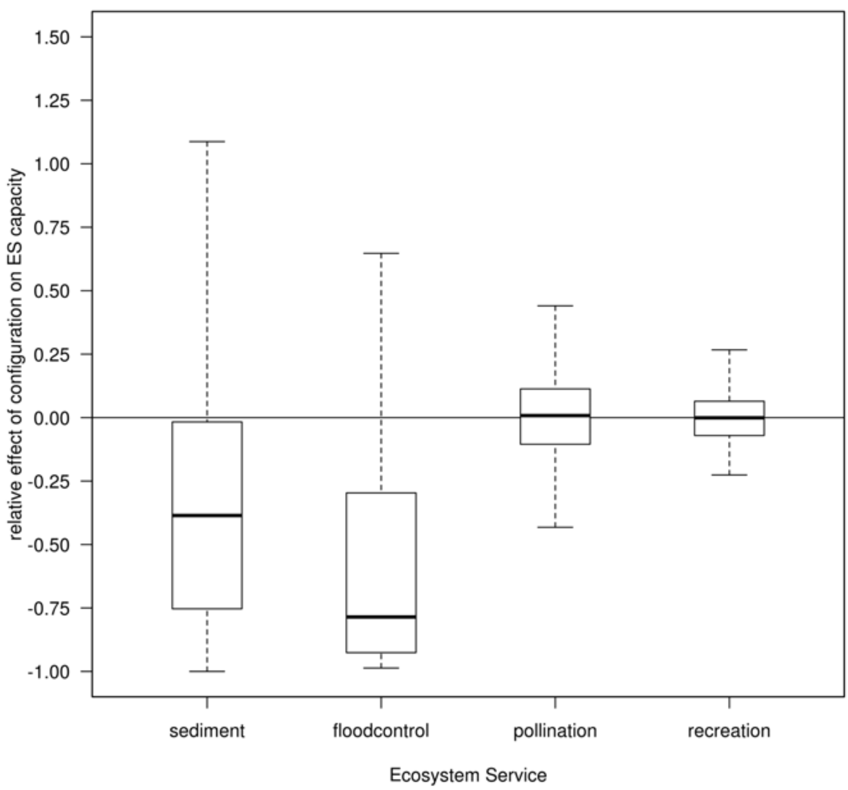
For all four ES mapped here, incorporating landscape configuration changes ES capacity compared to incorporating landscape composition alone. The ES capacity of all ESs differs between the composition and the configuration model although the effect depends on the level of aggregation (national, watershed and cell). Differences between the composition and configuration model are largest at the cell scale (Table 2.3). At the cell scale, flood control is affected most strongly by configuration, followed by sediment retention and pollination. Accounting for configuration does not have a strong effect on landscape aesthetics at the cell level. At the watershed scale differences between the composition and configuration model are smaller on average, but certain watersheds show large differences. Especially watersheds with little remaining natural vegetation are very sensitive to the spatial arrangement of natural vegetation. At the watershed scale, pollination is affected most strongly by configuration followed by flood control. Accounting for configuration has a small effect on ES capacity of sediment retention and landscape aesthetics. At the national scale differences between the composition and configuration models for ES capacity are small for all ESs (Table 2.3). In general, accounting for configuration makes substantial difference on the level of ES capacity at the cell scale and at the watershed scale. For the whole of Scotland local effects of configuration average out and in general hardly affect the ES capacity.

Table 2.3: Percentage mean absolute difference and standard deviation between total ES capacity for the composition and configuration model per ES. Results are presented at the national level, at the watershed level and at the cell level. Min-max present the minimum and maximum percentage difference for all watersheds or cells. A negative minimum value means that accounting for configuration results in a decrease in ES capacity for that watershed or cell.

		Mean Absolute Difference	SD	Min-max
Flood control	National	0.58%		
	Watershed	6.18%	9.34%	-82.67% - 296.65%
	Cell	122.8%	13.8%	-98.66% - 747.75%
Pollination	National	2.61%		
	Watershed	6.59%	11.75%	-83.01% - 22.92%
	Cell	13.8%	11.8%	-86.94% - 339.66%
Landscape Aesthetics	National	1.49%		
	Watershed	3.41%	2.81%	-22.41% - 16.23%
	Cell	7.68%	5.33%	-22.60% - 36.44%
Sediment retention	National	5.62%		
	Watershed	2.61%	5.59%	-59.16% - 35.05%
	Cell	49.9%	90.65%	-100% - 106.34%

At the cell scale, the relative effect of configuration on ES capacity differs per ES Figure 2.1). The relative effect of configuration to ES capacity shows a large range per ES, being both positive and negative for all ESs depending on location. Positive values indicate that the configuration model projects a higher ES capacity than the composition model for a given cell. Negative values indicate the opposite: the configuration model projects lower values of ES capacity than the composition model. There are some clear differences between ESs in the (relative effect of composition and configuration to ES capacity. For sediment retention and flood control, the relative effect of configuration is often larger and negative, resulting in a decrease in the mapped ES capacity. For pollination and landscape aesthetics, the relative contribution of configuration is often smaller and the contribution to ES capacity is evenly distributed between positive and negative effects. To conclude, accounting for configuration is likely to have a larger effect on mapping the ES capacity for flood control and sediment retention compared to pollination and landscape aesthetics at the cell scale. For most cells accounting for configuration will reduce the mapped ES capacity for flood control and sediment retention for the majority of cells.

Figure 2.1: Boxplots showing the relative effect of configuration to ES capacity at the cell level. All effects are scaled relative to the ES capacity value for the composition model (0 line). A value of 0.0 means that the outcome of the composition and configuration model are equal, whereas a value of 0.2 means that accounting for configuration results in a 20% increase in ES capacity, and vice versa. The solid black line in each boxplot represents the median effect. Outliers are not depicted. Boxplots, including the outliers are depicted in S1.



2.4.2 Spatial pattern of differences between composition and configuration

The differences between the composition and configuration models differs per location for all ESs but only shows a spatial pattern for landscape aesthetics for the whole of Scotland (Figure 2.2). Landscape aesthetics capacity decreases in areas with homogeneous land cover, being either agricultural-dominated or having large areas of natural land cover. Landscape aesthetics capacity increases at the edges of the agricultural areas, where the landscape aesthetics capacity of both agricultural and natural land cover is positively influenced by the diversity of land cover types in close proximity. For example, cropland areas are mainly located along the eastern shore of Scotland where large areas show a decrease in ES capacity. At the edge of these cropland areas landscape aesthetics capacity increases because of a mix of natural and artificial land cover types.

To illustrate the effect of configuration at smaller spatial scales we mapped the difference between the composition and configuration model within a single watershed (Figure 2.3). This particular watershed was selected because it has a gradient in dominant land cover type from dominant natural land cover in the northern part to dominant agricultural land cover in the southern part, and a mix of agricultural and natural land cover in the central part. Within this watershed the differences between the composition and configuration models show a spatial pattern for all ESs, resulting in areas with a predominant increase or decrease in ES capacity. The resulting spatial pattern differs per ESs. For flood control, the spatial pattern is similar throughout the watershed. Flood control capacity increases strongly along flow paths and closer to water courses, whereas it decreases in most cells located further away from water courses. For sediment retention, the spatial pattern of the differences between the ES models is dependent on the dominant land cover type. In areas dominated by agricultural land cover sediment retention capacity primarily decreases, while it increases for land cover adjacent to streams and rivers. Agricultural land cover has a low filtration capacity resulting in sediment accumulation towards the stream. In areas dominated by natural vegetation sediment retention capacity mostly increases, while it decreases for land cover closest and furthest away from streams and rivers. Many natural land cover types have a high filtration capacity meaning that most sediment is intercepted close to the source and there is no sediment accumulation towards the stream. Pollination capacity is only assessed for natural land cover in the agricultural-dominated area, because of the maximum distance of 500 meters between cropland and nesting sites. Pollination capacity predominantly shows little difference between the composition and configuration model. Increased pollination capacity is observed for smaller habitat patches and for edge habitats, whereas decreased pollination capacity is observed in the interior of larger habitat patches. Landscape aesthetics show a similar spatial pattern of the differences between the composition and configuration model within a watershed as at the national scale.

2.4.3 Landscape metrics and the difference between the composition and configuration model

Correlations between landscape metrics and the difference between the composition and configuration model range from the absence of any correlation at all radii for flood control and sediment retention to moderate correlations for pollination and landscape aesthetics (Table 2.4). Landscape aesthetics show a relatively high correlation with "patch density" and "land cover richness", especially at 250 meter radius. This correlation decreases with increasing radii. Pollination shows a correlation with "patch density" and "land cover richness" up to 500 meter. Flood control and sediment retention show no correlation with landscape metrics. "Distance to water", as a simple proxy for the flow accumulation, shows low correlation with flood control and no correlation with sediment retention.

Figure 2.2: Percentage difference between the composition and configuration model per configuration results in an decrease in ES capacity, and vice versa. The maps only depict the result for mainland Scotland, excluding all islands. For flood control, urban areas were assigned a value of zero for flood control potential in the composition.

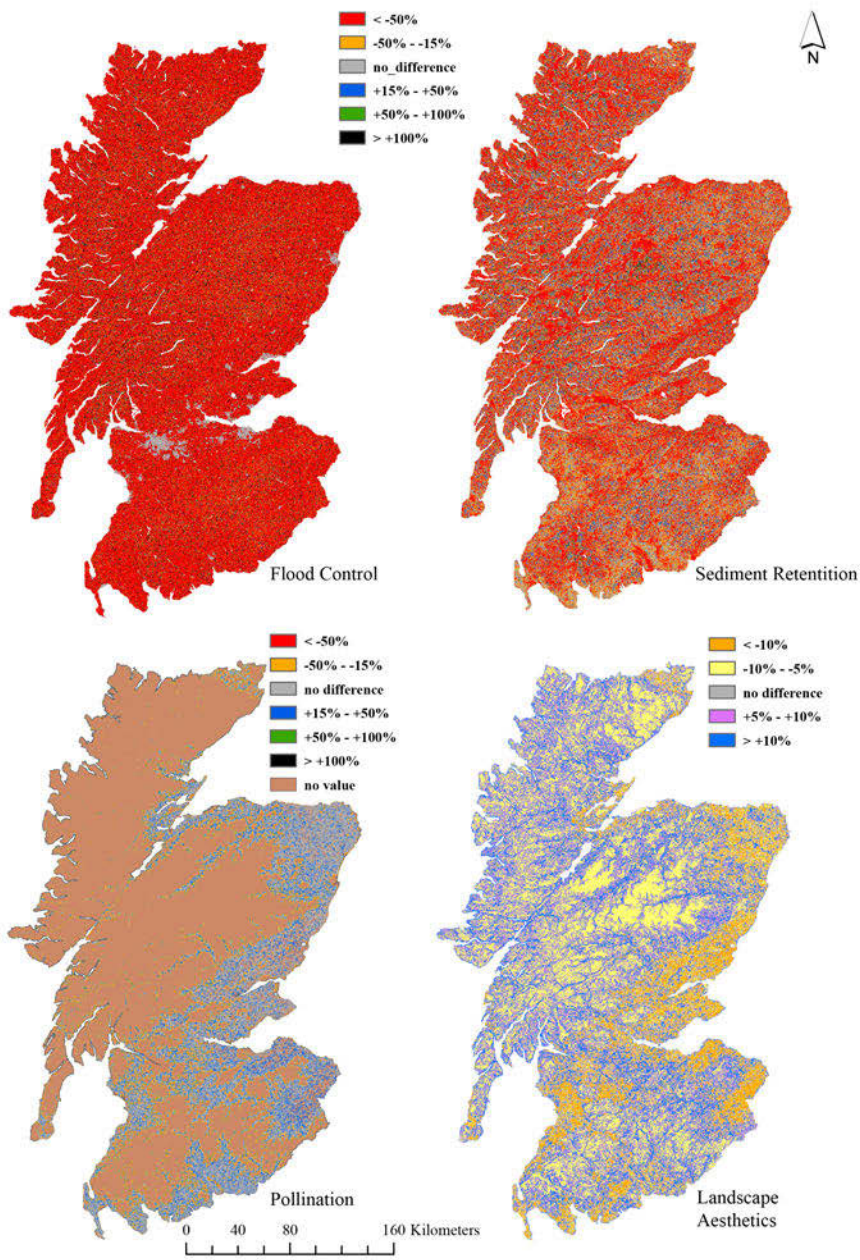
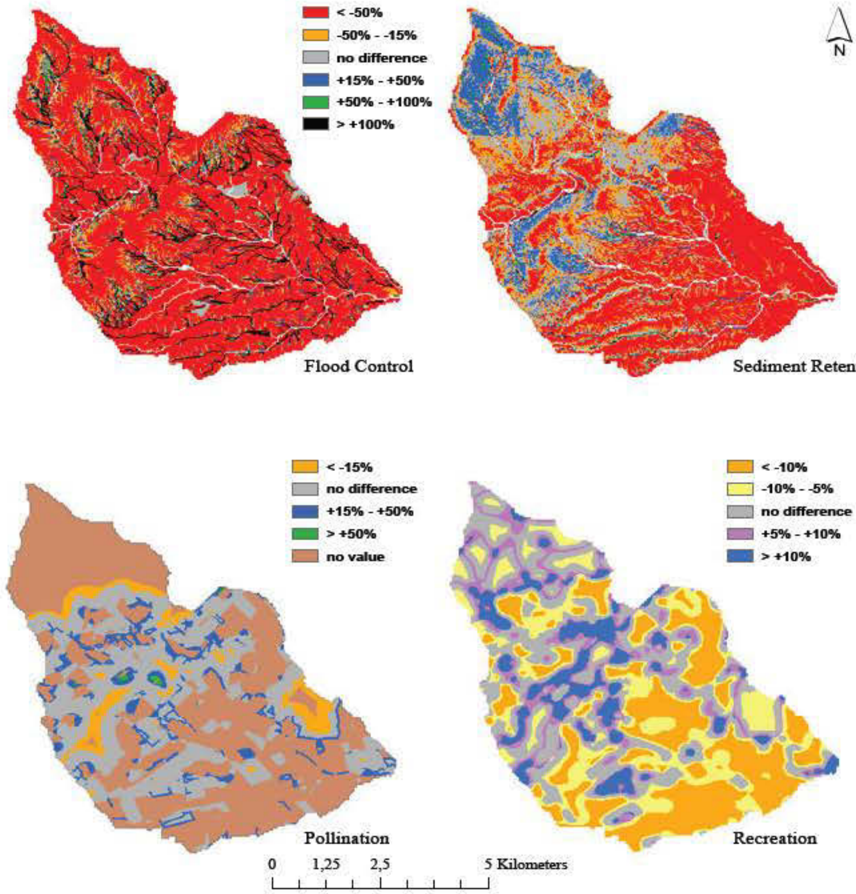


Figure 2.3: Percentage difference between the composition and configuration model per cell within a watershed. A percentage decrease means that accounting for configuration results in an decrease in ES capacity, and vice versa. For pollination, there are many cells with "no value" as we did not assign values for pollination potential for cropland cells and for habitat cells further than 500 meter away from cropland. This particular watershed was selected because it represents a gradient in dominant land cover type from natural dominated land cover in the north to agricultural dominated land cover in the south.



2.5 Discussion

2.5.1 Landscape configuration and ESs in field studies

We started our analysis by reviewing the empirical evidence for a relation between landscape configuration and ten ESs. We found evidence for a relation between landscape configuration and nutrient retention, pollination, landscape aesthetics and sediment retention. Moreover, there is evidence for a relation between landscape configuration and flood control, for unsaturated soils. The results from the review are likely to be applicable outside Scotland because our review incorporated studies from sites in temperate and continental climates.

Mitchell et al. (2013) performed a review on landscape connectivity and ESs, highlighting that there is a lack of empirical studies on the relation between landscape connectivity and ESs and only finding a clear relation between landscape connectivity and pollination. In line with the findings from Mitchell et al. (2013) we conclude that the number of studies that assess the relationships between landscape configuration and ESs remains limited and that empirical evidence for the relationship remains scarce. In contrast to their review we found evidence for a relation between landscape configuration and pollination as well as for four additional ESs. In our review we included a broader definition of configuration effects in relation to ES capacity and we included studies from outside the ES community, which could explain the difference between the two reviews.

Table 2.4: Pearson correlation coefficients for a set of landscape metrics and the difference between the composition and configuration model per ES. Landscape metrics were calculated for land cover within three radii around a cell (250, 500 and 1000 meters). For flood control and sediment retention the correlation is also calculated for the distance from the cell to the nearest stream or river in each watershed. All correlations, indicate by *, are statistically significant (p<=0.05).

	Radius(m)	Pollination	Landscape Aesthetics	Sediment Retention	Flood Control
% natural vegetation	*250*	0.05	0.36*	-0.01*	-0.01*
	500	-0.01	0.33*	-0.01*	-0.01*
	1000	-0.06	0.28*	0.00	-0.01*
LC richness	*250*	0.32*	0.74*	0.02*	0.03*
	500	0.32*	0.34*	0.00	0.01*
	1000	0.01	0.54*	0.01*	0.02*
Patch Density	*250*	0.29*	0.72*	0.02*	0.03*
	500	0.26*	0.54*	0.01*	0.02*
	1000	0.04	0.36*	0.00	0.01*
Distance to water				-0.03*	-0.10*

Our review also highlighted that different aspects of configuration affect ES capacity. In a recent conceptual paper, Mitchell et al. (2015) identified four possible ways in which landscape fragmentation affects ES capacity, namely increased interspersion of land cover, increased isolation, reduced patch size and increased edges. The first three aspects of the framework by Mitchell et al. (2015) would in our classification be grouped together under "configuration of multiple patches" and are expected to have an effect on pollination, landscape aesthetics, erosion control and nutrient retention. The fourth aspect, an increased amount of forest edges is according to our review expected to affect pollination. In addition to the framework suggested by Mitchell et al. (2015) we incorporated the specific location of land cover that affects flood control and linear elements that affect pollination, erosion control and nutrient retention.

2.5.2 Consequences for mapping ES capacity

Models and indicators for mapping ESs do not commonly account for configuration. Our results suggest that for particular ESs, when quantified at the watershed or cell scale, it is important to account for configuration. The effect of accounting for configuration changes with the resolution of the analysis. This has important consequences for mapping ESs. ES assessments interested in the quantification of the level of ES capacity at large, national, scales do not have to account for configuration effects as local effects of configuration largely average out. Although the total ES capacity at the national scale is hardly affected configuration does change the locations with higher and lower ES capacity at the national scale. At the watershed and cell scale, accounting for configuration can affect the level of ES capacity. Only for landscape aesthetics accounting for configuration had a small effect on the level of ES capacity at all scales.

In a hypothetical landscape, Mitchell et al. (2015a) showed that landscape fragmentation effects on ESs are non-linear from the cell to the landscape scale. The notion that the effects of configuration on ESs are scale dependent (Fahrig et al., 2011; Mitchell et al., 2015a), is confirmed by our results. In our models, configuration had a different effect on mapping ESs at the cell and the watershed scale. Flood control and sediment retention were strongly affected by configuration at the cell scale. Pollination was less strongly affected by configuration at the cell scale but accounting for configuration at the watershed scale had the strongest effect on pollination. For sediment retention local negative and positive effects of configuration tended to average out at the watershed scale. Mapping of flood control was still affected by configuration at the watershed scale. Our review highlighted that landscape configuration shapes erosion and runoff processes at the scale of individual hill slopes and watersheds. Assessment tools have been developed that can account for the different erosion and runoff processes at hill slope and watershed scale which could be implemented in ES mapping studies (Goodrich et al., 2011).

Accounting for configuration in ES mapping can partly address issues raised by previous researchers on mapping ESs using solely landscape composition. In the UK, Eigenbrod et al. (2010) showed that using landscape composition models to map ESs resulted in a mismatch with primary data on ESs. This mismatch is attributed to three types of generalization errors (Plummer, 2009), of which the uniformity error can be accounted for by configuration. The uniformity error is associated with the assumption that a land cover type supplies the same amount of ES capacity irrespective of for example patch size, management history or location in the landscape. ES mapping models that accounts for configuration can incorporate effects of patch size and location in the landscape, and could thus partly address the uniformity error.

We tested whether certain commonly-used landscape metrics can be used to proxy the configuration effect in mapping ESs. Only for landscape aesthetics, we found a correlation between the change in modelled ES capacity and the landscape metrics "land cover richness" and "patch density". The correlation diminished with increasing radii over which the landscape metrics were calculated. Care should be taken in interpretation and generalization of this result. For land cover richness the high correlation may be explained by the use of SHDI to account for land cover diversity in our configuration model. Moreover, at 250 meter radius "land cover richness" and "patch density" are highly correlated (0.95). Lastly, the correlation is likely highest at 250 meter radius, because we used a constant view shed of 200 meters in the model. Spatial autocorrelation in the landscape metrics can contribute to the correlations found at larger resolutions. Nonetheless the correlation between the two landscape metrics and the difference between the composition and configuration model for landscape aesthetics capacity is high, irrespective of the dominant land cover type. Our findings are in line with previous research finding opportunities to assess landscape aesthetics using SHDI and patch density for landscapes in Germany (Frank et al., 2013). "Land cover richness" and "patch density" within the direct surroundings of a cell are therefore likely to be appropriate proxies for the change in modelled landscape aesthetic capacity at the cell level due to configuration.

We found no or only very weak correlation between changes in modelled ES capacity and the tested landscape metrics for the other ESs. Previous research showed that landscape metrics could be used to account for the effect of changing landscape structure on ES supply after land cover change (Frank et al., 2012). Other landscape metrics, not tested here, could possibly explain some of the variation in the difference between the composition and configuration model. Nonetheless, the fact that sediment retention and flood control are not correlated to any of the landscape metrics tested and the distance to stream was surprising. In the analysis we compared configuration effects at cell level in very different types of watersheds, both in size and in landscape composition, because we were interested in the use of landscape metrics for mapping ES at large spatial scales. For landscape aesthetics we did partly control for differences in landscape composition by assigning different landscape preferences scores to depending on the dominant land cover type. We did not test whether landscape metrics could better explain the configuration effect at the cell level within a single watershed or for similar landscape compositions. However, previous research on the correlation between landscape metrics and sediment retention showed that the results were dependent on the land cover map used (Uuemaa et al., 2005) highlighting that caution should be applied in using landscape metrics to account for configuration effects. Alternatively, to capture the complex and variable effect of configuration on these ESs we suggest to map ES capacity using spatially explicit modelling frameworks that account for configuration (e.g. Nelson et al. 2009; Jackson et al. 2013).

2.5.3 Mapping approach

The ES models in this paper have not been developed for the purpose of most accurately mapping ES capacity, but rather to allow for the distillation of a configuration effect. We next discuss our mapping approach with respect to the capacity of distilling the configuration effect.

Large outliers were observed for the flood control and sediment retention model at the cell scale (S1). These outliers may be explained by two factors. First, the sediment retention model has been calibrated at the watershed scale, meaning that possible errors at the cell scale have not been checked. Second, we accounted for spatial pattern on flood control using flow accumulation. Although flow accumulation has been used by previous researchers (Chan et al., 2006) a more detailed hydrological model might render different results. However, we aimed to rely on existing techniques for mapping ESs and therefore applied flow accumulation. InVEST, a commonly-used tool to map ESs across larger scales, only accounts for the land cover capacity to retain incoming precipitation and does not account for the spatial configuration of the watershed by accounting for water input from upstream sites (Sharp et al., 2015). Third, neither the flood control model nor the sediment retention model accounts for saturation. This is a common issue in ES models (Nelson et al., 2009), but some modelling tools do account for the effect of soil saturation on water holding capacity and consequently on flood control capacity (e.g. Laterra et al. 2012; Jackson et al. 2013). The most recent version of the InVEST model for fresh water provisioning now also accounts for water holding capacity (Sharp et al., 2015). Incorporating saturation in ES models is important for mapping ES capacity influenced by configuration. The ES capacity at a site does not only depend on the characteristics of the cell but also on the input of either water or sediment from upstream sites. Not accounting for saturation may only result in a possible overestimation of ES capacity and is unlikely to change our finding that accounting for configuration predominantly leads to a reduction in mapped ES capacity per cell.

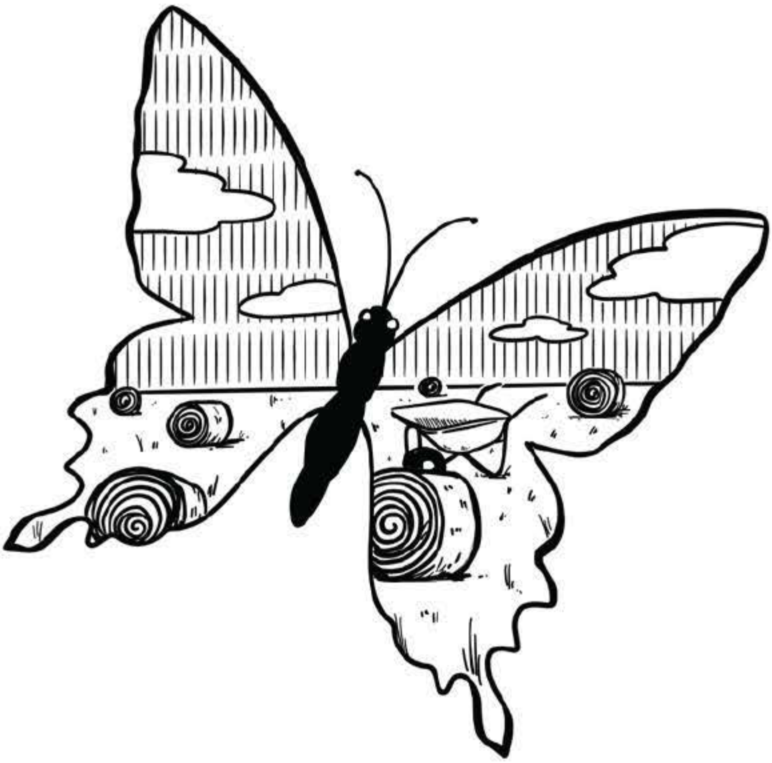
A limitation of our mapping approach is that the configuration models did not account for linear elements, which might result in an underestimation of the effect of configuration to ES capacity. Based on the literature review, linear elements affect ES capacity of flood control, pollination and nutrient and sediment retention. Maps of linear elements across scales are being developed using observation data or processing high resolution imagery data (e.g. Aksoy et al. 2010; van der Zanden et al. 2013). At the European scale linear elements have been incorporated in the assessment of soil erosion (Panagos et al., 2015) and pollination (Schulp et al., 2014b). Although linear elements only covered less than 5% of the agricultural area, their presence increased the visitation probability of pollinators by 5% to 20% (Schulp et al., 2014b). Linear elements were the sole source of pollination capacity in 12% of the agricultural areas (Schulp et al., 2014b). Both mapping approaches rely on the density and not the location of linear elements within an area. Readily available, full coverage data on the location of linear elements at a resolution of 1km or smaller are however not yet available, despite efforts in this direction for the UK and Scotland (Barr and Gillespie 2000; Brown et al. 2014). Riparian habitats are another example of important landscape features capable of providing multiple ESs (Jones et al., 2010). The area of riparian forest is suggested as main indicator to map ES capacity for nutrient retention at the European scale (Maes et al., 2016). For the UK, the combination of land cover data at 25-meter resolution with flow accumulation maps provides opportunities for the assessment of ESs in riparian zones and the impact of management and land cover change in riparian zones on future ES supply (Jones et al., 2010).

2.5.4 Implications for landscape management

Is landscape heterogeneity good? An important question to assess the promises of multifunctional landscapes is whether there is a uniform response of ESs to configuration (Macfadyen et al., 2012; Jones et al., 2013). Our results suggest a non-uniform response of ESs to configuration. First of all, based on the results of our review not all ESs have a clear relation to configuration and for those ESs that are affected by configuration, different aspects of configuration affect the ESs. The effect of configuration on ES capacity is thus dependent on the ES under study. Landscape configuration and heterogeneity are believed to be capable of alleviating trade-offs between ESs. Our results suggest that configuration acts in different ways on different ES and is thus likely to introduce new trade-offs between ESs. Effects of landscape configuration on single and multiple ESs will likely depend on the location, the composition and configuration of the landscape, the set of ES studied and the level of aggregation in ES assessment. Second, we also highlighted non-uniform responses in the effect of configuration on mapping ES capacity at the cell and watershed scale (Mitchell et al., 2015a). The total ES capacity at the watershed scale can be only slightly affected by configuration, but the locations with high ES capacity can change strongly after accounting for configuration. Management interested in maintaining locations of high ES supply should therefore account for configuration to effectively identify priority areas.

In our models we looked at the effect of configuration on single ESs and did not look at the effect of configuration on the capacity of multiple ESs. Landscape heterogeneity is not only expected to affect the level and location of ES capacity, but also the interactions between ESs (Bennett et al., 2009). In our literature review we did encounter studies on the relation between landscape configuration and the capacity of multiple ESs, but in line with findings from a previous review (Mitchell et al., 2013), none of the studies looked at interactions between multiple ESs. In ES mapping studies, multifunctional landscapes or ESs hotspots are often identified by combining individual ES maps for an area (e.g. Qiu & Turner, 2013). Combining individual ES maps cannot be used as a way to reveal possible relations between configuration and landscape heterogeneity when interactions between ESs are not accounted for. Future empirical research on multifunctional landscapes should focus on the interactions between ESs and the effect of configuration on these interactions. Importantly, interactions between two ESs can be affected by landscape configuration, even when the ES capacity of the individual ESs is not affected by configuration. Landscape management for multifunctional landscapes should account for landscape configuration on ES supply as well as on the interactions between ESs.

Our results also provide relevant input for landscape management aimed at optimizing ESs. It has been argued that ES management can use landscape structure to enhance ES capacity (Jones et al., 2013; Turner et al., 2013). Our results provide tangible evidence that tools for managing ESs in landscapes should account for landscape configuration both in assessments of ES capacity given current land use, and in the assessment of impacts of land cover change (Lautenbach et al., 2011). Care should however be taken by translating findings from ES mapping studies to landscape management, especially when accounting for configuration. One reason is that recommendations based on mapping studies using coarse land cover maps, such as CORINE, are likely to only have limited use in explaining patterns of biodiversity and ESs for landscape management (Gimona et al., 2009). A second important aspect is that not only ES capacity but also ES demand is likely affected by landscape configuration. Studies on aesthetics and recreation often account for the landscape configuration in demand parameters such as accessibility (Guo et al., 2001; Chan et al., 2006; Chen et al., 2009; Larondelle and Haase, 2013; Nahuelhual et al., 2013) or visitation rates (Wood et al., 2013). Other research has more strongly focused on the effects of landscape configuration on ES flows arguing that ES flows are more strongly impacted than ES capacity (Mitchell et al., 2015b, 2015a). The fact that landscape configuration will affect ES capacity, ES demand and ES flows of multiple ESs simultaneously provides both opportunities and challenges for optimizing ESs through landscape management. Nonetheless, accounting for configuration can help protect ES capacity through identification of priority areas, and can help optimize ES capacity in landscapes through spatially explicit land management. The large differences between the composition and configuration model at the cell scale, but the smaller differences in ES capacity at the watershed scale, suggest that there is room for optimizing ES capacity by explicitly accounting for configuration in landscape management.



3. Management intensity and the trade-off between production and biodiversity

Abstract

Land-use intensification is increasingly being used to boost agricultural production and is recognized as a major threat to biodiversity. However, little is known about the simultaneous effects of land-use intensification on biodiversity and yield. To determine the responses of species richness and yield to intensification, we conducted a global meta-analysis synthesizing 115 studies which collected data for both variables at the same locations. We extracted 449 cases that cover a variety of areas used for agricultural (food, fodder) and silvicultural (wood) production. We found that, across all production systems, intensification is successful in increasing yield (grand mean +20.3%), but it also results in a loss of species richness (-8.9%). The larger the steps of land-use intensification, the greater are the gains in yield while losses in species richness persist. For example, intensification from low to medium, increased yield by 6% and from low to high by 28.8%. Simultaneously, species richness was reduced by 7.7% and 12.1% respectively. Small intensification steps within low intensity systems did not affect yield (-0.7%) or species richness (-0.8%), while within high-intensity systems species loss (-6.1%) and yields gains (+15.2%) were detected. Intensifying within medium intensity systems revealed the highest yield increase (+84.9%) and showed the largest loss in species richness (-22.9%). Production types differed substantially in their magnitude of richness response, with silvicultural systems showing the smallest (-1.6%) and crop systems the largest losses (-21.2%). Across all sub-analyses, the unexplained variation remained high, which underlines a lack of quantitative studies that simultaneously measure richness and yield. These finding suggest that, in many cases, land-use intensification drives the trade-off between species richness and production, even in already intensively used areas. As such, this global synthesis highlights, that the increase of agricultural yields through land-use intensification might hamper achieving global targets of preserving life on land.

This chapter is published as:
Michael Beckmann, Katharina Gerstner, Morodoluwa Akin-Fajiye, Silvia Ceaușu, Stephan Kambach, Nicole L. Kinlock, Helen R. P. Phillips, **Willem Verhagen**, Jessica Gurevitch, Stefan Klotz, Tim Newbold, Peter H. Verburg, Marten Winter, Ralf Seppelt. (2019). Conventional land-use intensification decreases species richness and increases production: A global meta-analysis. Global Change Biology. https://doi.org/10.1111/gcb.14606

The main author of this article is Michael Beckmann. Willem Verhagen is a co-author on this publication and has contributed to the overall idea for the paper, the setup of the coding protocol, the selection and coding of the articles, the interpretation of results and the writing of the paper.

3.1 Introduction

While some human-managed lands can provide benefits for the protection of individual species (e.g. Loos et al., 2014), the need to use land to produce food and other goods is generally at odds with biodiversity conservation (Green et al., 2005; Foley et al., 2011; McShane et al., 2011; Cardinale et al., 2012). Today, the majority of Earth's land surface has been transformed by human activities and is subject to some kind of human land use, such as agriculture, settlements, infrastructure but also mineral extraction (Hooke et al., 2012). There is indication that land conversion has slowed down while the production of renewable resources continues to increase (Seppelt et al., 2016). This raises concern that – besides land conversion – land-use intensification poses a major threat to biodiversity (Pereira et al., 2010; Maxwell et al., 2016), as changes in land-use intensification typically result in a loss of species richness (Gerstner et al., 2014; Kehoe et al., 2015; Newbold et al., 2015), the most widely reported measure of biodiversity (Isbell et al., 2011). Although the importance of land use for biodiversity and the provision of goods has been widely acknowledged in conceptual (e.g. Clough et al., 2011; Tscharntke et al., 2012; Fischer et al., 2017; Seppelt et al., 2017) and empirical studies (Gerstner et al., 2014), relatively few studies have measured the simultaneous effects of land-use intensification on species richness and yield in the same locations.

Recent scientific debates on closing yield gaps or conserving biodiversity in agroecosystems have addressed the effects of land use either on agricultural production *or* biodiversity conservation (e.g. Mauser et al., 2015; Newbold et al., 2015; but see Denmead et al., 2017; Garibaldi et al., 2017; Egli et al., 2018). A notable exception trying to include both perspectives is the land sharing-sparing concept (Chappell and LaValle, 2011; Phalan et al., 2011b, 2011a). However, although it has sparked various discussions among the scientific community, the concept has been criticized for being overly simplistic and lacking applicability to many real landscapes (Cunningham et al., 2013; Fischer et al., 2014; von Wehrden et al., 2014). In order to better inform the highly relevant debates about trade-offs between agricultural production and biodiversity, a quantitative synthesis that addresses the effect of land-use intensification on the species richness-production relationship is clearly needed.

Land-use intensification generally aims at increasing production and/or financial benefits through creating more output from the same area. The type, extent and intensity of land use vary considerably and are highly dependent on biophysical conditions, national priorities, policies, local needs as well as the availability of technologies and knowledge (Václavík et al., 2013; van Asselen and Verburg, 2013). Land-use intensity can range from slight alterations in management practices to a substantial reshaping of landscapes; it can involve small increases in manual labour but also the use of large machinery, whilst potentially making use of natural products for fertilization and pest control or the broad-scale application of chemicals for the same purposes. Such different intensification steps may be called "conventional", "organic" or "nature friendly", labels that can have different meanings depending on the location (Seufert et al., 2017). Here, we focus on agricultural and silvicultural land use intensification in general, which we define as changes in management practices that aim to increase production on already used land (Box 3.1).

Studies addressing the effect of intensification on species richness and production on continental or global scales often incorporate data generated by models or country-scale statistics (Kehoe et al., 2015; Delzeit et al., 2017). While there have been numerous studies collecting field data on both agricultural or silvicultural production and species richness within a defined area (Gabriel et al., 2013; Norvez et al., 2013), a global analysis synthesizing such data is still lacking. It remains, for example, unclear whether a steady increase in yield and decrease in species richness along a gradient of land-use intensification can be found, whether both can be increased at the same time or whether smaller decreases in species richness for a given increase in yield are possible (Fischer et al., 2014; Seppelt et al., 2016). Understanding changes in species richness within production systems is important aside from conservation concerns. Species support key ecosystem functions and services within agricultural landscapes (e.g. Klein et al., 2003), although the details of these relationships still remain unresolved in many (Isbell et al., 2017) but not all cases (Seabloom et al., 2017).

Consequently, we here investigate the three-way relationship between intensification, species richness and yield, by synthesizing the published literature that collected these data in the same locations. With this meta-analysis we further try to identify whether a general trade-off between species richness and yield is detectable and if there is evidence for situations in which yield can be increased with simultaneous positive or neutral effects on species richness. To quantitatively compare studies along a gradient of land-use intensification, we use a general classification of land-use intensification. We categorized land-use intensification steps that are comparable across different landscapes globally and between different production systems (wood, green fodder, crops), and that take into account the initial land-use intensity and the magnitude of intensification (Box 3.1). This approach allows for the inclusion of specific land-use strategies (e.g. conventional, organic or nature friendly). We focus on production-species richness trade-offs, but exclude other aspects of the multifaceted food-security and sustainability debates (e.g. long-term yield stability, economic profits; (Fischer et al., 2017a; German et al., 2017; Seppelt et al., 2017)). To unpack the various facets of the intensification-species richness-production relationship, we structure this meta-analysis to highlight the following contrasts in examining impacts on each of them:

I) Land-use intensification out of low-intensity systems;
II) Land-use intensification towards high-intensity systems; and
III) Land-use intensification in gradual, small steps.

We synthesized studies that collected data on species richness and agricultural yield (biomass per unit area or the nearest available proxy) in response to land-use intensification. A study had to measure both response variables within or adjacent to the same spatial units of study to be included in this meta-analysis. As the effects of land-use intensification on species richness may depend on taxa, product type, land-use history and climate (Ellis et al., 2013; Gerstner et al., 2014; Newbold et al., 2015; Perring et al., 2016), we investigated whether the relationship between species richness and yield is influenced by these factors. Furthermore, we checked if the results were robust across different units of yield, harvested crop species, if species richness and yield were measured from the same species group or if data was collected at different plot sizes.

3.2 Material and Methods

3.2.1 Literature search and screening protocol

We conducted a systematic review in compliance with the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) framework (Moher et al., 2009) (S3). We searched the Web of Science database for search terms related to land use, biodiversity and yield (see S4 for the full search term and all refinement options employed). We included all articles published since January 1, 1990 in English or Spanish. The final search resulted in 9,909 studies.

We included studies meeting the following selection criteria: Studies had to measure both species richness and yield in the same site in response to the application of land-use intensification. This way, we included both, studies that measured the effect of intensification over time (i.e. using the pre-intensified state as control and the measurements after intensification as treatment response) and studies that measured several sites in response to different intensities (i.e. space-for-time substitutes). Out of the full initial set of papers, we manually screened the abstracts of 6,116 studies and retained studies only if they contained

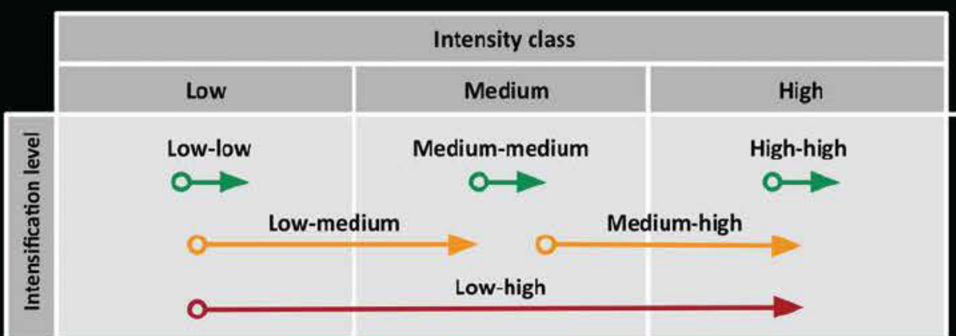
Box 3.1: Framework used for the identification of land-use intensity classes and intensification levels.

We defined **land-use intensity** based on labour input, input of fertilizer and/or pesticides, harvest and other aspects. For studying a gradient of land use intensity we define three broad classes of land use intensity.

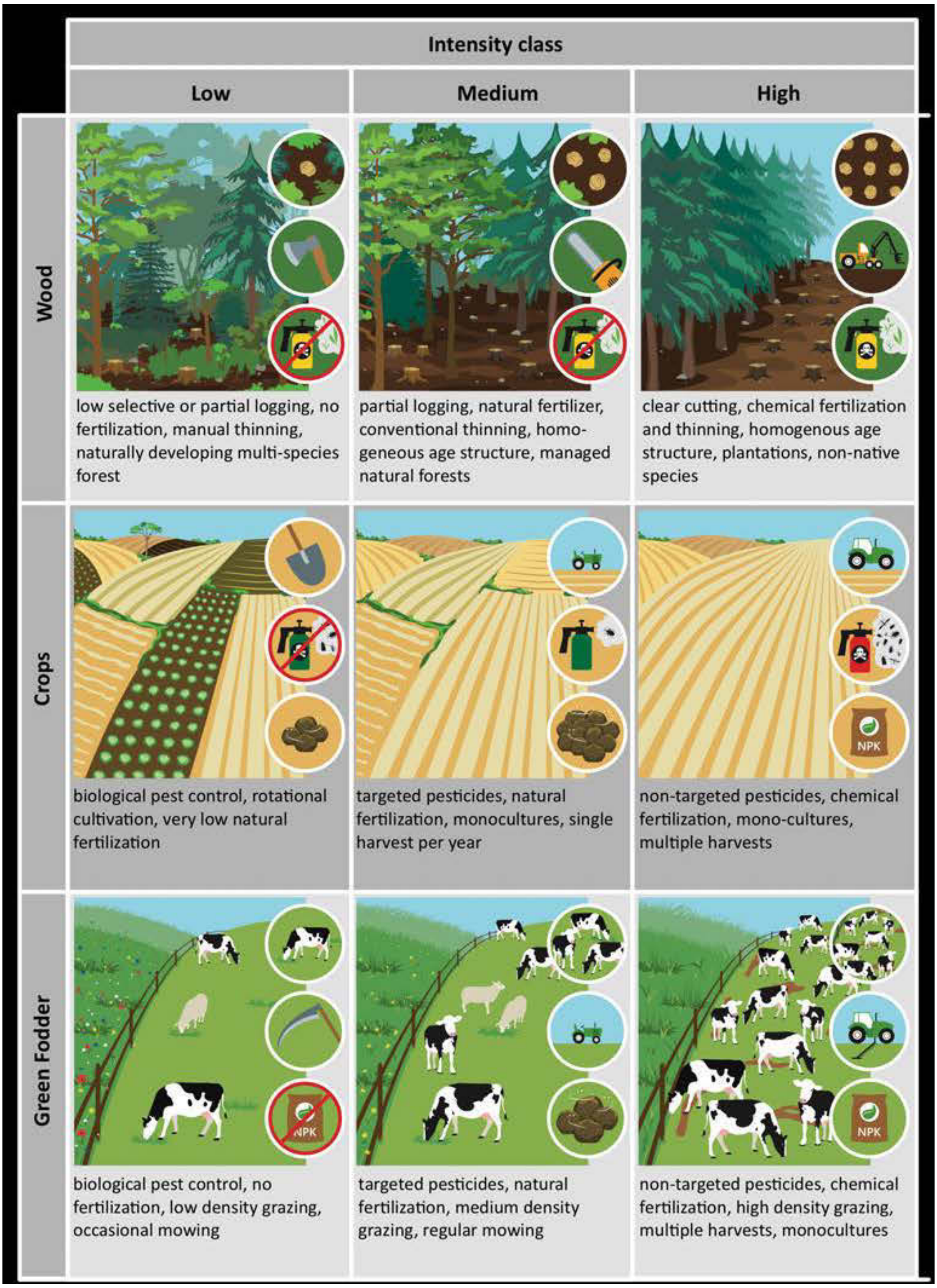
This broad categorization of land use intensity can be characterized separately for the three **production systems** "crops", "wood", "green fodder". Figure C (next page) illustrates this and list specific aspect of the land-use intensity for each production system. This type of classification of land-use intensification allows for comparisons across production systems and regions (Hudson et al., 2014).

By this we can distinguish different degrees of **land-use intensification.** Land-use intensification could occur "in small steps" by slightly increasing the pre-existing management activities (e.g. more fertilization, higher intensity of thinning), which defines land-use intensification categories low-low, medium-medium, high-high, see Figure B, green arrows). Second, land use-intensification can move the system into another intensity class (e.g. low-medium, medium-high, low-high) by a more substantial change in management (e.g. introducing broad spectrum chemical pesticides, clear cutting instead of partial logging).

	Intensity class		
	Low	Medium	High
Labour	Manual	Small machinery	Large machinery
Fertilization	No very low organic	Organic or chemical	High input chemical
Pest control	No or biological pest control	Targeted pesticides	Non-targeted pesticides
Harvest	Manual, low frequency	Small machinery, medium frequency	Large machinery, high frequency
Other	Rotational cultivation, low density grazing	Monocultures, medium density grazing	Monocultures, high density grazing

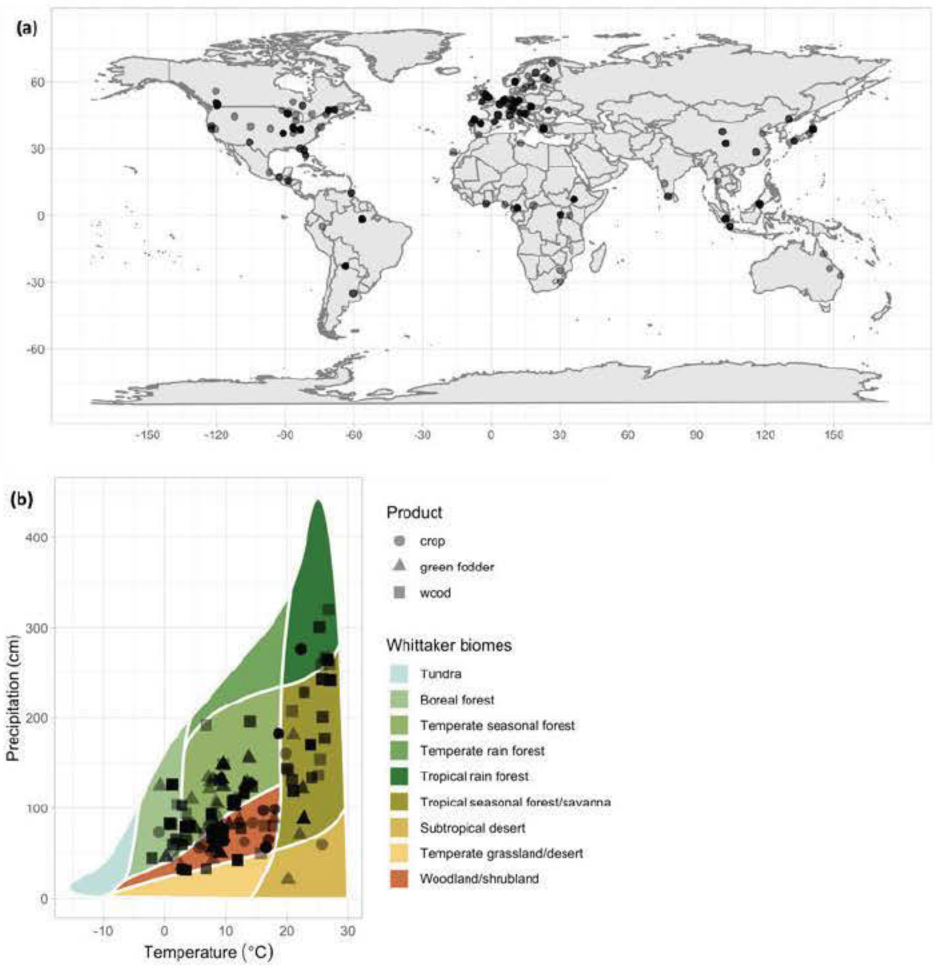


Box 3.1(continued)



information about land use, species richness, and/or yield. In order to filter the remaining 3,793 studies, we used a machine-learning algorithm based on ensembles of Support Vector Machines (SVMs) developed for systematic reviews of the medical literature (Wallace et al., 2010). The machine-learning algorithm correctly identified 84% of the manually screened studies as being relevant, with a specificity of 51% (standard deviation 0.016), i.e. the model eliminated half of the irrelevant. The full text documents of all studies identified as potentially relevant (1371), both screened manually or through machine learning, were acquired and processed further (S3, S7).

Figure 3.1: Locations of sites included in the meta-analysis. (a) Sites of the 449 cases (292 for species richness and 157 for yield) that were extracted from 115 studies (see S13 for a complete list of references). At each site data on species richness and yield in response to conventional land-use intensification was collected. (b) Illustrates the distribution of sites and cases across climate zones in a Whitacker plot. If several cases were located at the same sites, the points are overlaid and thus darker.



3.2.2 Data extraction and validation

From the initial 1,371 studies, 115 studies had sufficient data to be included (see Figure 3.1 for a global distribution of the studies). Means, standard deviations and sample sizes for control (lower land-use intensity) and treatment (higher land-use intensity) were extracted from the text, tables or figures using ImageJ (Schneider et al., 2012). If data were not completely available in the main document and the supplementary material, we requested them from the corresponding author. Studies that did not report means or sample sizes were excluded from the analysis. This resulted in a total of 115 studies that were used in subsequent analyses (see S15 for the full list of references). Data coding and data review were undertaken by eight of the co-authors. Initially, studies were coded as a group to assure inter-coder consistency and reliability. Subsequently, frequent internal reviews were conducted to maintain consistency. Each document has at least been coded by two of the co-authors.

We distinguished between production systems for crops, green fodder and wood and within each system we defined land-use intensity and its steps of intensification (Box 3.1 and S5 for details of the classification approach). We used a classification system for land use intensity based on a defined set of management practices (Box 3.1). This conceptualization of intensification is based on energy use and labour (i.e. more intensive systems use more energy inputs), but it does not classify land use intensity based on a quantification of energy use. Within each study, we differentiated comparisons based on different land-use intensification steps, taxa, or product types, resulting in a total of 449 cases of which 292 cases are for species richness and 157 for yield. Unequal numbers of cases for species richness and yield are caused by studies that have gathered species richness data on different taxa but report only one type of yield.

Biodiversity was quantified using species richness (i.e. numbers of species), as reported by the authors. When species abundances were provided, species richness was calculated as the total number of species with at least one recorded individual. Species were grouped into three groups of taxa: vertebrates, invertebrates, and plants.

Yield was most commonly reported as a mass-per-area (e.g. tons per hectare), or volume-per-area (e.g. cubic meter of timber per hectare). All products were assigned to one of the three product-types: crops, green fodder, and wood. We always coded the provided measure of yield that was as close as possible to the final product (i.e. if a study on cacao plantations reported annual cacao harvest and wood volume of the cacao trees, we included only the cacao yield). Multiple crops on the same area or multiple harvests per year were treated individually and coded as separate cases. Throughout all studies in green fodder and crop system and for many of studies in forest systems it was possible use yield change under intensification as biomass or volume per area as effect size.

For approximately two thirds of the forest studies yield was not reported in mass per area or volume per area units. Here, we used the nearest available information given by the authors of the study on standing biomass of commercially relevant trees such as basal area or total volume of standing biomass (area-per-area measurements). Although these measures are proxies, they have previously shown to be reliable predictors for harvest yields of many commercial tree species. Although more complex models are suggested, Junior et al., (2014) show that basal area already explains 96% of the variability in estimating above ground biomass ($r^2 = 0.968$). Especially as we here focus on relative yields change, we expect deviation due to nonlinearities to be considerable small. Consequently, we tested for dependence of unit of measure to ensure comparability of coded data (S13). Additionally, in forest systems the final harvested amount of wood was only rarely available directly in the studies. According to the information provided by the authors we carefully interpreted the given change in yield. Internal reviewing of these critical cases was used to verify the interpretation of results.

The distribution of crop species planted per land-use intensification class was analysed to support interpretation of relative yield change. For each crop species, data on the nutritional value was obtained from USDA Nutrient database (protein, fat and sugar content for a given yield change, S14).

In order to test whether effects of land-use intensification varied according to the environmental context, we assigned each study location to one of five climate zones according to the Köppen-Geiger classification (Kottek et al., 2006): tropical climate; arid climate; temperate climate; cold, continental climate; polar climate (see S6 for details).

In order to analyse each study location according to their land-use history (i.e. length of human land use at this location) we developed a classification to represent five main land-use history classes characterized by major developments in agriculture and silviculture (Vasey, 1992; Mazoyer and Roudart, 2006): Origin of agriculture; Expansion of agriculture; Middle Ages; Modern agriculture and Green Revolution (S8 for details). We applied these classes to a global dataset dating back to 5950 B.C. (KK10 dataset; (Ellis et al., 2013)) which describes the proportion of land within 0.1°× 0.1° grid cells that has been used by humans in time steps of 50 years. For each study case, we extracted the date of first significant use (defined as 20% of human-used area within a grid cell).

3.2.3 Data analysis and statistical methods

Using the extracted means, standard deviations and sample sizes for both lower-intensity control and higher-intensity treatment, we calculated log-transformed response ratios and variances (Koricheva et al., 2013). The response ratio can be interpreted as the species richness or yield of the higher-intensity treatment as a proportion of that in the lower-intensity control. Hence, a response ratio of 1.0 signifies no change; i.e., species richness or the yield are the same after intensification, and, for example, a value of 0.8 indicates 80% of the species or yield remains after intensification (i.e. 20% loss). Log-transformed response ratios were used in the analyses but were back-transformed and converted to percentage change for ease of interpretation in the results presented.

We imputed missing data for standard deviations (169 out of 449 cases) as suggested by Gerstner et al., (2017) based on predictive mean matching using the R package mice (version 2.22; (Buuren and Groothuis-Oudshoorn, 2011)). The relationship between observed means of response ratios, standard deviations and number of samples was first fitted to the subset of data without missing values. Multiple imputation chains were then generated using Gibbs sampling, i.e. a random draw from the posterior predictive distribution of model coefficients. We imputed missing standard deviation values using the mean of 50 imputation chains.

We analysed variation in species richness and yield effect sizes using linear mixed-effects meta-analysis models (R version 3.0.1: function rma.mv, package metafor version 1.9.8; (Viechtbauer, 2010)). This function is particularly designed for performing multilevel meta-analyses. We used restricted maximum likelihood to estimate mean effect sizes and their variances and maximum likelihood estimation to compare the goodness-of-fit between models. The models tested are specified in the caption of Table 3.1.

We accounted for (1) non-independence of observations from the same study, and (2) non-independence from relatedness of multiple intensification steps within one study by specifying covariances between effect sizes X and Y as,

$$cov(X,Y) = cor(X,Y) * sqrt(Var(X)) * sqrt(Var(Y)),$$

where $cov(X,Y)$ is set to 0.5 if X and Y belong to the same study and share a control or treatment, because effect size X determines 50% of effect size Y and vice versa. All models were fitted using study and case nested within study as random effects to account for dependencies of multiple outcomes within the same study (Nakagawa and Santos, 2012). The covariates "land-use intensity step", "species group", "product", "main climate zone" and "land-use history" were fitted as fixed effects.

We compared three models for species richness and three models for yield, using different sets of covariates (Table 3.1): (i) a model containing "land-use intensification step" as a single explanatory factor; (ii) a model that additionally contained "species group" and "product" (for yield the model contained "product" only) and their interactions with "land-use intensification step"; and (iii) a model that additionally include "land-use history" and "climate" and their interaction with "land-use intensification step". We evaluated the goodness-of-fit of the models using various statistics provided by the R-package metafor since there is no consensus on a single best fit statistic: *AICc*, and *BIC* as measures of overall model fit, the model heterogeneity Q_M (Hedges and Olkin, 1984) and its *p*-value of statistical significance, the unexplained (or sampling) heterogeneity Q_E, and the proportion of observed variance explained by the model, calculated as the ratio of Q_M to $Q_T = Q_M+Q_E$. The ratio is comparable to the R^2 value from linear regressions but uses the ratio of weighted sums of squares. Finally, we provide I^2 as a measure of the amount of heterogeneity within studies (I^2(Study ID)) and within study cases (I^2(Study Case)) relative to the total heterogeneity (Nakagawa and Santos, 2012). For the models of yield, species group is not considered a relevant explanatory variable and is therefore not included (Koricheva et al., 2013).

We compared mean percentage change of species richness and yield predicted by the models. Mean effects of land-use intensification were considered significant if their 95% confidence intervals (CIs) did not cross zero. To test pairwise differences of factor-level effects for land-use history and climate, we averaged model predictions of the full model (containing all covariates) across land-use intensification steps, species groups, and products and performed pairwise *t*-tests with the Holm-correction for multiple comparisons (S9). If distributions of effect sizes within groups are normal, both tests (pairwise *t*-test and boxplot) result in the same conclusions (Crawley, 2007). If distributions are skewed, however, conclusions may differ.

We explored possible correlated or confounded variables in our dataset, including 1) measuring species richness and yield on the same organism group (for example, in grassland systems where species richness and yield may both be derived from the same plants); 2) direct linkage of yield to land-use intensity (e.g. through harvesting techniques such as clear-cuts or selective logging); 3) measures of yield expressed in very different terms; or 4) the dependence of species richness on spatial scale.

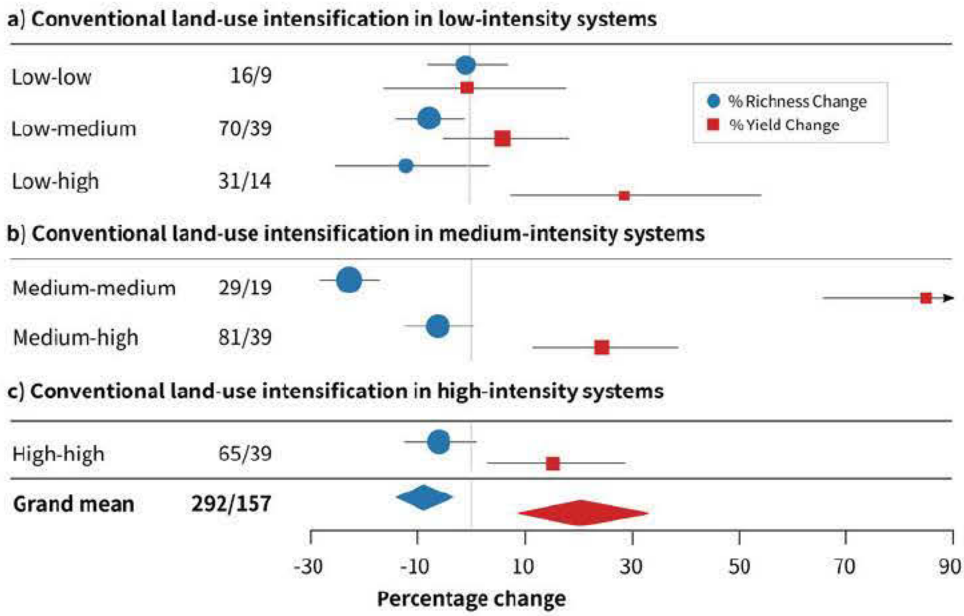
All code, performing the analysis as described in the Methods and all underlying data is available at GitHub: https://github.com/KatharinaGerstner/LUBDES_MA.

3.3 Results

3.3.1 General effect of land-use intensification

When considering all possible intensification steps, product types and species groups together, we found that land-use intensification leads to a significant overall gain in yield (+20.3% [95% confidence interval: +8.9, +33.0]), and significant loss of species richness (-8.9% [-14.0, -3.5]; grand mean in Figure 3.2). None of the intensification steps provide a statistically significant indication that yields and species richness could be increased at the same time. Situations, in which intensification increases yield but with no significant effect (although with negative mean values) on species richness, were identified: intensification within the high-intensity land-use class (Figure 3.2; species richness: -6.1% [-12.5, +0.8], yield: +15.2% [+3.1, +28.7]), medium to high intensification (species richness: -6.3 [-12.3, +0.2], yield: +24.3% [+11.6, +38.5]) and low to high (species richness: -12.1% [-25.2, +3.4], yield: +28.8% [+7.5, +54.3]). It is important to note that these results exhibit heterogeneity among studies (Table 3.1), with a range of impacts in individual studies on both species richness and yield.

Figure 3.2: Change in species richness and yield as a result of conventional land-use intensification. Mean percentage change in species richness and yield to conventional intensification steps (1st column). The number of samples for species richness/yield cases is given in the 2nd column. Numbers of studies from which these cases were extracted are given in Appendix S8. Error bars and horizontal points of the diamonds show 95% confidence intervals. The arrow denotes a confidence interval larger than axes. Effect sizes were calculated and analysed using log response-ratios, which were back-transformed and converted to percentage change. Results shown are based on 449 cases and are derived from the full models as shown in Table 3.1a and 1b.



3.3.2 Land-use intensification in low-intensity systems

Increasing intensification of low-intensity systems resulted in strong positive yield responses and negative responses to species richness with increasing variation between studies. Small intensification efforts in low intensity systems (e.g. a low increase of stocking density in extensive grasslands) did not show any clear effect on yield or species richness (Figure 3.2). A further intensification (from low to medium intensity, e.g. introducing low-input fertilization in a pasture system) resulted in negative effects on species richness (-7.7% [-13.7, -1.3]) without benefitting yields on average (+6.0% [-5.0, +18.3]). When increasing land use from low-intensity systems to become a high-intensity system, the mean effect on species richness was negative although non-significant (-12.1% [-25.2, +3.4]), and there was a significant positive effect on yield (+28.8% [+7.5, +54.3]), although heterogeneity among studies was high.

3.3.3 Land-use intensification towards high-intensity systems

Intensification towards high-intensity system showed negative but not significant effects on species richness, while yield increases were significant but declined with larger intensification steps. When a low-intensity system was transformed to a high-intensity system (e.g. changing a manually worked field to a highly mechanized agricultural system), the relative gains in yield were highest (low-high; +28.8% [+7.5, +54.3];Figure 3.2). Yield gains were lower when intensification was carried out from medium to high intensity (medium-high; +24.3% [+11.6, +38.5]), or where there is already high-intensity usage (high-high; +15.2% [+3.1, +28.7]). While species richness showed no significant response to intensification in the three analysed intensification steps that end in high land use intensity, we did find strong negative trends.

Table 3.1: Goodness-of-fit statistics for meta-analysis models. (**a**) species richness, (**b**) Yield. Abbreviations: *ΔAICc* = Akaikes' Information Criterion and *ΔBIC* = Bayesian Information Criterion expressed as the difference of each model compared with the best-fitting model; Q_M = model heterogeneity; Q_E = unexplained (or sampling) heterogeneity; p(Q_M) = proportion of observed variance explained by the model calculated as the ratio of Q_M to $Q_T = Q_M + Q_E$. See Appendix S4 for more details.

a) Species richness (n = 292 cases)

	ΔAICc	*ΔBIC*	Q_M	p(Q_M)	Q_E	R^2	I^2 (Study ID)	I^2 (Study Case)
Intercept only								
	639.72	445.04	9.97	0.002	9321.49	0.001	0.81	0.19
Land-use intensification step								
	425.80	249.08	233.61	<0.001	9204.25	0.025	0.7	0.24
Land-use intensification step + species group + product type								
	86.784	0.00	636.24	<0.001	7022.39	0.083	0.67	0.33
Land-use intensification step + species group + product type + climate + land-use history								
	0.000	13.21	830.35	<0.001	5674.32	0.128	0.75	0.25

b) Yield (n = 157 cases)

	ΔAICc	*ΔBIC*	Q_M	p(Q_M)	Q_E	R^2	I^2 (Study ID)	I^2 (Study Case)
Intercept only								
	3670.74	3572.33	13.13	<0.001	10794.13	0.001	1.00	1.5E-07
Land-use intensification step								
	899.51	815.56	2794.80	<0.001	8899.40	0.239	1.00	2.5E-07
Land-use intensification step + product type								
	562.23	509.75	3161.94	<0.001	6646.92	0.322	0.48	5.3E-01
Land-use intensification step + product type + climate + land-use history								
	0.000	0.000	3863.68	<0.001	2780.89	0.581	0.18	8.2E-01

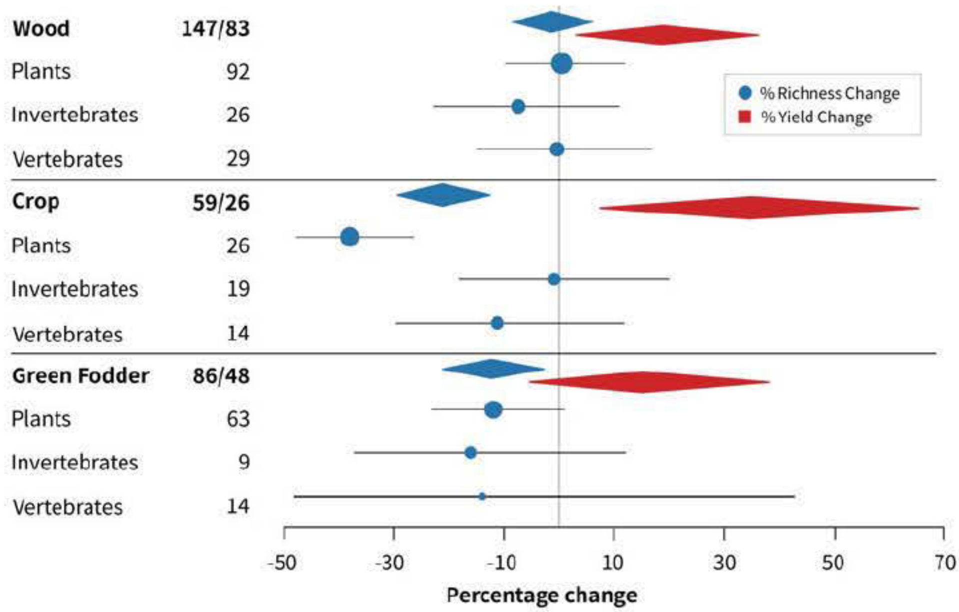
3.3.4 Land-use intensification in small steps

Intensification within low-intensity systems (low-low) showed neither a positive effect on yield nor negative effects on species richness (Figure 2c, S10). Surprisingly, intensification within medium-intensity systems (medium-medium) was associated with the most pronounced increases in yields (+84.9% [+65.8, +106.1]) and greatest losses of species richness (-22.9% [-28.1, -17.4]). Intensification within systems already at high intensity (high-high) results in smaller, yet significant, increases in yield, while there is a negative, but not significant effect on species richness (Figure 2; S10).

3.3.5 Species groups, product types and other covariates

Overall, animal species were not significantly affected by higher land-use intensity while plants were (invertebrates -6.7% [-17.2, +5.0]; vertebrates -2.9% [-14.4, +10.2]; plants -11.4% [-17.8, -4.5], Figure 3.3). Species richness decreased most (-21.2% [-29.9, -11.5]) and production increased most (+33.3% [+7.4, +65.4]) with intensification in crop-production systems. Green fodder systems showed similar, but not significant, trends (species richness: -12.4% [-21.8, -1.9], yield: +14.2% [-5.6, +38.2]), whereas in wood-production systems species richness did not respond to intensification (-1.6% [-8.8, +6.2], Figure 3.3) though yield increased by 18.6% ([+3.0, +36.6]). Loss of species richness varied depending on the history of land use but showed no trend (S11). Species richness declined most and yields increased least in arid climates, while in the tropics, species richness declined substantially and yields increased relatively little (S12).

Figure 3.3: The effect of conventional land-use intensification on species richness and yield, analysed by product type and species group. Mean percentage change in species richness and yield in response to conventional land-use intensification, for different species groups and product types (1st column). The number of samples (species richness/yield) is shown in the 2nd column. For each species group and product type, the mean across all intensification steps is shown. The impact of species group on yields was not tested. Error bars and horizontal points of the diamonds show 95% confidence intervals. Effect sizes were calculated and analysed as log response-ratios, which were back-transformed and converted to percentage change here. Results shown are based on 449 cases and are derived from the full models as shown in Table 3.1a and b.



The tested covariates explained a significant proportion of the heterogeneity (Q_M) in effect sizes for both species richness and yield ($p(Q_M) < 0.05$; Table 3.1). Furthermore, all models that included these covariates showed lower *AICc*, *BIC*, and increased R^2 compared to the null model without covariates. The goodness-of-fit statistic *AICc* suggested that for both species richness and yield the model incorporating all covariates was the most parsimonious model. Furthermore, the heterogeneity statistic, $Q_{M,}$ suggests that a significant amount of heterogeneity was explained in these full models as well. We found that mean effect sizes did not differ depending on whether species richness and yield were measured from the same species group (t = -0.196, df = 136.85, p = 0.845). We found a significant difference between linked and unlinked yield and land-use intensity measures only for wood products (t = -2.38, df = 42.5, p = 0.022). Pairwise *t*-tests showed no differences in the effect size for different yield units (Mass/area-Area/area p = 0.2, Count/area-Area/area p = 0.12, Count/area-Mass/area p = 0.37). As the scale dependency of species richness is a well-known constraint for interpreting species richness data in meta-analyses (Chase & Knight, 2013), we tested for scale dependency using the reported size of sampling area. Linear regression of the mean effect size for species richness as a function of log-transformed sampling area did not reveal a significant relationship ($F_{1,271}$ = 0.027, p = 0.869; all results are shown in S13). The distribution of crop types and major nutrient contents varied considerably across intensification steps, thereby revealing and unbalances representation within agricultural production systems, S14).

3.4 Discussion

With this global meta-analysis, we find that there is, on average, a trade-off whereby increases in agricultural/silvicultural yields are accompanied by decreases in species richness (grand mean in Figure 3.2). When breaking down these results by the magnitude of intensification steps, species groups and product types, we find that intensification of land use is often successful in increasing yield, while in many situations, it results in a significant loss of species richness. Neither in the grand mean, nor in any sub-group analysis (Figure 3.2;Figure 3.3) we could identify situations in which intensification increases yield and provides benefits for biodiversity at the same time. However, uncovering such, often discussed, 'win-win' situations is likely to require more holistic approaches that include socio-economic variables (Batáry et al., 2017; Hanspach et al., 2017). Notwithstanding, while individual examples certainly support the idea that biodiversity can benefit from intensification (Pywell et al., 2015; Katayama, 2016), this meta-analysis finds no support for this currently being a common and generalizable pattern.

However, we are able to identify situations in which yield can be increased with smaller (i.e. non-significant) losses of species richness, analogously denoted as 'win-no-harm' situation. First, species richness in wood production systems shows little to no response to land-use intensification (Storkey et al., 2015; Thomas, 2015), which might be explained by long harvest cycles and the lower disturbance over time needed to manage forests (Paillet et al., 2010). Second, animals are not as negatively affected by intensification, as are plants. This might reflect differences in the overall mobility of some species groups, which possibly mediates the impacts of intensification (Tscharntke et al., 2005). We found no trade-off between production and species richness within low-intensity systems (low-low), where neither yield nor species richness showed notable responses to intensification. If intensification steps remain small, these systems might hold some potential for ecological intensification (Geertsema et al., 2016).

Surprisingly, intensification within high intensity systems (high-high) revealed losses of species richness, highlighting that even high intensity systems harbour species that will be lost through further intensification. The proportionally lower yield gains within high intensity systems compared to other intensification steps (e.g. low-high, medium-high), indicates that high intensity systems are reaching production limits (Seppelt et al., 2016). On the contrary, intensification within medium-intensity systems (medium-medium) provides the greatest increase in yields, but is also accompanied by the highest loss of species richness. Consequently, these systems might be the first choice if seeking maximum production increases, but they are also most vulnerable to species richness decline. Here, the apparent context-dependency of the effects land-use intensification has on species richness indicates that great caution must be taken when interpreting the outcomes of this meta-analysis.

Studies on the biodiversity-productivity-relationship have provided evidence that higher biodiversity leads to higher ecosystem productivity (Seabloom et al., 2017). These outcomes have often been utilized to conclude that biodiversity also benefits the production of biomass actually consumed by humans (Liang et al., 2016). However, such approaches make the rather unrealistic assumption that people do not care which kind of biomass is harvested, even though most products depend on single species. This needs to be considered carefully when drawing conclusions on the benefits ecosystem productivity provides for humans. While in some cases biodiversity may help to support yield (Klein et al., 2003), our results challenge the idea that a positive biodiversity-primary-production relationship directly translates into a benefit for biomass actually appropriated by humans. However, given that the underlying data is unbalanced for the major crops and nutrient types (S14), the collected results for agricultural systems should be treated carefully.

This meta-analysis is no exception to often encountered shortcomings when dealing with the synthesis of data on a global scale (Gerstner et al., 2017). By focusing on species richness, which is the most widely reported measure of biodiversity, we use an incomplete measure of biodiversity (Pereira et al., 2013), ignoring homogenization effects and the hidden loss of rare or endemic species. Therefore, important negative consequences of land-use intensification might be masked. However, too few studies reported on more robust measures of biodiversity (e.g. species abundances, Shannon diversity), alongside yield responses to permit the inclusion such metrics in this meta-analysis. Furthermore, by using a space-for-time substitution approach in this meta-analysis, other detrimental effects of land-use on species richness might have been hidden as well (Berg et al., 2015; Elmendorf et al., 2015; França et al., 2016). In addition, response ratios capture only the relative effects of intensification on species richness and yield. This way, changes in absolute values or species identity might be obscured. Finally, although we found no effect of spatial grain of the study sites, we cannot entirely rule out that the dependence of species richness on spatial scale does not affect the outcomes presented here.

A clear caveat to the implications of this meta-analysis for policy or management is that one size does not fit all: in all analyses, the heterogeneity among studies was large. Even where the statistical models explained significant amounts of variation, individual cases may exhibit different outcomes. Identifying the nuances and complexities that make up the intensification-species-richness-production relationship requires a solid foundation of data collected in a globally representative number of different production systems and species groups as suggested by German et al. (2017). As this synthesis has shown, only a comparatively low number of studies have done this so far. Instead, the majority of previous research has focused on the effects of land-use intensification either on biodiversity *or* on yields (S7; (Mauser et al., 2015; Newbold et al., 2015)).There were not enough studies that would explain some of the uncovered heterogeneity and allow drawing conclusions on a finer scale (e.g. for species-product-climate combinations). It becomes clear that a greater number of studies should aim to gather both types of information in the future. One way out of this predicament would be the establishment of global, long-term research networks such as has been done with the Nutrient Network (NutNet; (Stokstad, 2011)).

In a world where human requirements almost always outweigh conservation objectives, one of the major challenges is to identify the form and location of land-use intensification that will best preserve the biodiversity, ecosystem functions, and ecosystem services upon which agricultural production ultimately depends. It is crucial that future studies focus more on areas already used for agriculture or silviculture as these harbour a substantial amount of species which will be lost through intensification. Given the predicted increases in the human population and consumption, it is likely that used land will be intensified further in the future. It is also likely that even low-intensity systems, such as smallholder farms, which account for more than 50% of agricultural land globally (Graeub et al., 2016), will turn to intensification in order to boost yields. We show here that the scientific community knows far too little about the simultaneous effects of land-use intensification on biodiversity and yield to provide well-founded support for policy and management. Therefore, we strongly argue for future studies to continue the search for pathways of ecological land-use intensification (Geertsema et al., 2016; German et al., 2017). These should bring together the perspectives of farmers, foresters and conservationists by accounting for socio-economic and biophysical aspects and quantify biodiversity as well as yields.

4. Identifying spatial priorities for ecosystem services across Europe: the importance of accounting for ES demand and ES flow

Abstract

Policies and research increasingly focus on the protection of ecosystem services (ESs) through priority areas. Priority areas for ESs should be identified based on both ES capacity and ES demand while accounting for the flow zones between areas of ES capacity and ES demand. Here we tested ways to account for ES demand and flow zones to identify priority areas in the European Union.

We mapped the capacity and demand of a global flow (carbon sequestration), a regional flow (flood regulation) and three local flow services (air quality, pollination and urban leisure). We used Zonation software to identify priority areas for ESs based on six experiments. First, we developed two experiments with and without accounting for ES demand. Second, we developed four experiments to test for the effect of accounting for the ES flow zone.

There was only 37.1% overlap between the top 25% priority areas with and without accounting for ES demand. Moreover, the level of ES maintained in the priority areas clearly increased after accounting for ES demand, especially for ESs with a small flow zone. Accounting for the flow zone had a smaller effect on the location of priority areas and level of ES maintained, but resulted in a more even distribution of ESs maintained across the flow zones. Accounting for demand and flow zones primarily enhanced the representation and distribution of ESs with local to regional flow zones, without strong trade-offs for the global flow ES.

Our results clearly showed that ignoring ES demand leads to the identification of priority areas in remote regions where benefits from ES capacity to society are small. Incorporating ESs in conservation planning should therefore always account for ES demand to identify an effective priority network for ESs.

This chapter is published as:
Verhagen, W., Kukkala, A.S., Moilanen, A., van Teeffelen, A.J.A. and Verburg, P.H. (2017), Use of demand for and spatial flow of ecosystem services to identify priority areas. Conservation Biology, 31: 860-871. doi:10.1111/cobi.12872

4.1 Introduction

Conservation planning increasingly incorporates ecosystem services (ESs) alongside biodiversity (Luck et al., 2012; Cimon-Morin et al., 2013). Conservation planning for ESs is often implemented through land management targets, including Aichi target 11 which aims to conserve 17% of the land for biodiversity and ESs (Convention on Biological Diversity, 2010). As land is scarce and funding is limited it is necessary to identify an efficient and effective network of priority areas for ESs. Spatial conservation prioritization provides the tools to do so (Moilanen et al., 2009). While developed for identifying biodiversity priority areas the approach has also been used for ES prioritization (Chan et al., 2006; Casalegno et al., 2014; Cimon-Morin et al., 2014; Schröter et al., 2014b). There are however differences in prioritizing areas for ESs compared to biodiversity (Luck et al., 2012). ESs are the benefits humans obtain from nature (MEA, 2005). Ecosystem functions and processes only become ESs when there is a demand for the service (Fisher et al., 2009). In setting priority areas for ESs one not only needs to account for the capacity of an ecosystem to provide a service, but also for the spatial variation in ES demand (Wolff et al., 2015).Studies on ES prioritization need to directly link demand and supply at a location.

However, areas with high ES capacity and the location of human beneficiaries do not necessarily coincide. The spatial connections between areas of ES capacity and areas of ES demand are called ES flow (Fisher et al., 2009). ES flow links areas of ES capacity and demand and ranges from global to local. Habitat for pollinators needs to be protected close to croplands whereas forests sequestering carbon can be conserved anywhere. For each ES it is possible to identify the flow zone, the area over which ES capacity and demand can be spatially linked. Priority areas for ES, therefore, need to be distributed across flow zones in order to be effective. Identifying priority areas for ESs thus needs to account for i) the spatial variation in ES demand and ii) the ES flow zone.

Current studies on prioritization for ESs do not always account for the spatial variation in ES demand. Previous research found large differences between priority areas with and without accounting for demand (Cimon-Morin et al., 2014). However, in their study areas of ES demand and ES capacity did not have to overlap, meaning that priority areas could have high ES demand and low ES capacity or vice versa.

Only very few studies have accounted for the flow zone of ESs. Orsi et al. (2011) assessed wood production by incorporating travel distance between communities and forests, while Chan et al. (2006) accounted for flow zones by assigning flood protection targets per catchments or recreation targets per city. To our knowledge, no study has quantitatively assessed approaches for ES prioritization combining ES demand and the distribution of ES.

This study aims to quantify the importance of accounting for demand and flow zones of ESs in identifying priority areas. Specifically asking: How is the spatial allocation of priority areas for ESs, and the level of ESs contained within top-priority areas, affected by: 1) accounting for the level of ES demand; and 2) accounting for the flow zone of individual ES? We address these questions using the European Union (EU) as study area. At the EU level policies are developed for protecting and enhancing ESs, such as the EU Biodiversity Strategy 2020 and the Strategy on Green Infrastructure (European Commission 2011; European Commission 2013). The EU Biodiversity Strategy aims to halt the loss of ESs, but actions towards this have mostly focused on ES capacity without accounting for the actual use or demand for ESs (Maes et al., 2016). Testing the effect of (not) considering demand for ESs and flow zones in identifying priority areas is relevant to the effective implementation of these types of policies.

4.2 Materials and methods

We used four regulating and one cultural ES for which both ES capacity and demand maps were available for the EU at a 1km resolution (Table 4.1). The ESs encompassed global (carbon), regional (flood control) and local (pollination, air quality, and urban leisure) ES flows. We included carbon sequestration as an example of a global flow ES to test whether focusing on more localized ESs resulted in a trade-off with maintaining carbon sequestration. We mapped the landscape's capacity to provide ESs (termed 'ES capacity') and the portion of ES capacity demanded by society, derived by combining ES demand and ES capacity maps, as explained below. The ES capacity and ES demanded capacity data were used as input to the prioritization analysis (Figure 4.1). To test the effect of accounting for flow zones we used either a single map per ES (EU), a map per administrative unit per ES (NUTS, an EU administrative unit) or a map per ES flow zone (FLOW) in the prioritization analysis (Figure 4.1).

Table 4.1: Summary of the ecosystem services and their flow zones used in the analysis.

	Source	Spatial flow	Flow zone	# of flow zones	flow zone area, median and range (km²)
Air Quality	(Maes et al., 2015b),	Local	EU city and commuting zone	436	773 (11–17470)
Carbon Sequestration	(Schulp et al., 2008)	Global	EU	1	NA
Flood Regulation	(Stürck et al., 2014)	Regional	(sub-)catchments >2km²	3878	250 (2-21574)
Pollination	(Schulp et al., 2014b)	Local	10x10 km area	37194	100 (100-100)
Urban Leisure	(S18)	Local	EU city plus 8 km buffer zone	538	767 (302-5363)

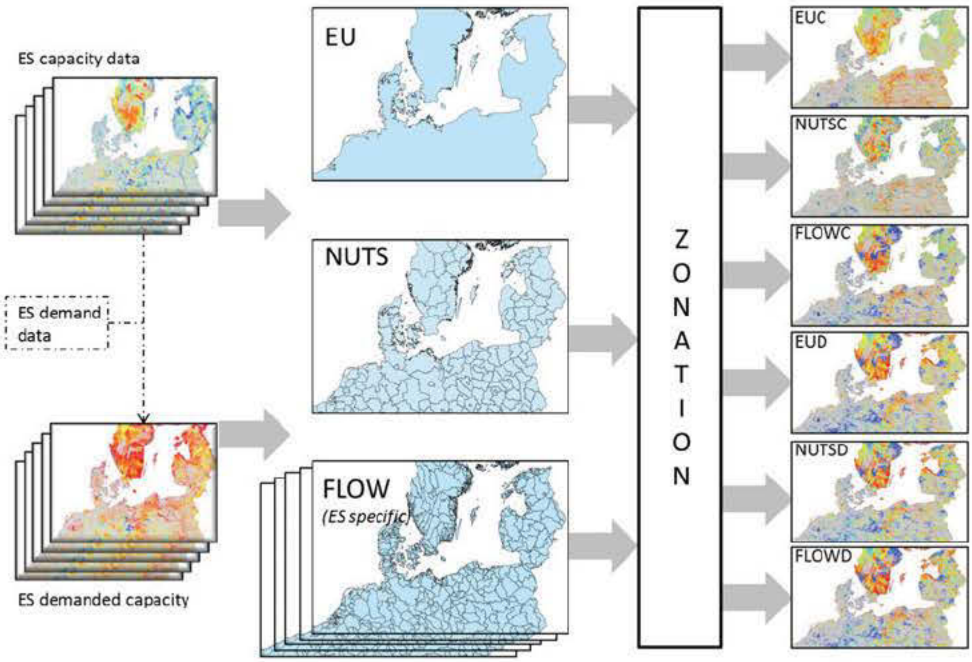
4.2.1 ES data and flow zones

In this section we describe for each ES the data used for ES capacity, ES demand, the calculation of ES capacity demanded and the delineation of flow zones for the prioritization analysis. The use of ES flow zones is described in section 'Prioritization analysis'.

Carbon sequestration capacity maps were derived from Schulp et al. (2008), whom used a bookkeeping approach combining below ground carbon sequestration for all land cover types with above ground carbon sequestration in forests (Tg C y^{-1}). We used the carbon sequestration map for the year 2000 and set negative carbon sequestration values to zero. As fulfilling the demand for carbon sequestration is not spatially constrained, we considered carbon sequestration a global flow ES. Hence, c*arbon sequestration capacity demanded* was set equal to ES capacity. The flow zone of carbon sequestration was the entire EU.

Flood regulation capacity and demand maps were derived from Stürck et al. (2014), whom assessed flood regulation capacity based on land cover, catchment type, precipitation, catchment zone, water holding capacity and land use and management. Catchment were delineated using an European catchment map (EEA, 2008) where catchments are delineated based on a DEM, landscape stratification and coastline data. The resulting catchments are Strahler order 5 catchments, often representing (sub-)catchments. We used potential flood damage, calculated using the damage scanner model of Bubeck et al. (2011) as proxy for flood regulation demand. Damage was calculated using land cover specific damage curves (€/ha) for a 50-year flood inundation level. Damages were aggregated per (sub-)catchment. Then the aggregated downstream demand for flood regulation per catchment was divided by the area of the upstream catchments which can potentially provide flood regulation (Stürck et al., 2014). This approach accounted for all benefits downstream of a (sub-)catchment. Flood regulation demand values were normalized between 0–1. We calculated the portion of the *flood regulation* *capacity demanded* by multiplying the ES capacity per cell, with the normalized ES demand per catchment. For flow scenarios (Figure 4.1), we used (sub-)catchments resulting in 3,878 flow zones (Table 4.1).

Figure 4.1: Schematic overview of the setup of our priorization of ecosystem services (ESs). Two sets of ES input maps are split into 3 different sets of spatial delineations. This results in a single map per ES for the EU, per NUTS (EU administrative unit), or per flow zone (an area with a unique demand–supply combination dependent on the ES flow) used as an input to the prioritization analysis. The 6 tests (EUC, NUTSC, FLOWC, EUD, NUTSD, and FLOWD) are described in section 4.2.2. Examples of priority areas are depicted on the left and right maps where red is low priority and blue is high priority.



Air quality regulation capacity and demand maps were derived from Maes et al. (2015) for NO_2 emissions. Air quality capacity was quantified using deposition velocity (m/s), mainly determined by the leaf area of plants (Derkzen et al. 2015). NO_2-emissions originate from fuel combustion by transport and industries. We used modeled NO_2-concentrations (ug/m3) as a proxy for *air quality regulation demand* assuming that demand is higher in locations with higher air pollution (Pistocchi et al., 2011). The deposition velocity and NO_2-concentrations were transformed to the same units (T/km^2/year). We calculated the portion of *air quality capacity demanded* by multiplying the NO_2-concentrations and the deposition velocity for each cell. For flow scenarios (Figure 4.1), we used city's functional urban area delineations (GISCO 2011), which includes the urban area and its commuting zone, resulting in 436 flow zones.

Pollination capacity and demand maps were derived from Schulp et al. (2014). Pollination capacity was mapped using the potential wild bee habitat per cell, assessed based on a reclassification of land cover and hedgerow density. Pollination demand was a combination of the pollination dependency of a crop type (0-100%) and the share of that crop type within a 1km^2 cell. We reassigned demand for pollination to cells with natural vegetation directly adjacent to croplands using the Moore neighborhood. We normalized the pollination demand between 0-1 and combined it with pollination capacity. The flow zone for pollination is limited by the pollinators flight distance. Most farmers have some opportunity to redistribute crops between fields and, thus, to redistribute benefits from pollination within a larger area. Redistribution of crops is likely to remain local but no information exists on the extent to which farmers can redistribute crops. For flow scenarios (Figure 4.1) we used a 10x10km zone for pollination, assuming that farmers have some opportunity to redistribute crops within this area. Only zones containing pollination dependent crops were included resulting in 37,194 flow zones.

Urban leisure opportunities was mapped using a combination of land cover data, distance to coasts, forest location characteristics and agricultural landscape structure (S16). We used population density in urban areas as proxy for demand. Following Paracchini et al. (2014), for each cell we calculated the aggregated urban population density within an 8km radius. We normalized the population density data between 0–1 and multiplied it with the ES capacity. The population density data are skewed, with some cells having very high population values. Therefore, we first winsorized the demand values based on the 95th percentile, i.e. assigning the 95th percentile value to all locations with values higher than the 95th percentile. For flow scenarios (Figure 4.1), we delineated European cities including an 8 km buffer, resulting in 538 flow zones.

4.2.2 Prioritization approach

We prioritized areas with natural vegetation for ESs in the EU, thus excluding the land cover classes ‘urban’ and ‘water’. Agricultural land was also excluded, except when hedgerows were present (van der Zanden et al., 2013). Croatia, Cyprus and Malta were excluded because not all ES maps covered these countries.

We used the Zonation spatial prioritization software version 4 to identify priority areas (Moilanen et al., 2005, 2014; Lehtomäki and Moilanen, 2013). Zonation was previously applied to identify priority areas for biodiversity (Pouzols et al., 2014) and ESs (Casalegno et al., 2014; Durán et al., 2014). Zonation produces a hierarchical prioritization of the whole landscape for multiple features (here ESs) simultaneously, based on weights and local occurrence levels of features, by iteratively ranking spatial units (here grid cells) minimizing aggregate loss of conservation value across features at each step (Lehtomäki and Moilanen, 2013). It is important that a balance between features is maintained over the prioritization (Moilanen et al., 2014).

When applying Zonation, one needs to choose how Zonation aggregates value over many partially conflicting feature layers (a.k.a. the cell removal rule). We used the Additive benefit function (ABF), which sums the loss of value across features converted via feature-specific benefit functions. The ABF calculates value based on all features co-occurring at a location and thus gives comparatively high priority to areas that can cost-effectively cover features simultaneously (Moilanen et al. 2014).

We developed six experiments that differ in how they account for ES demand and flow zones (Table 4.2). First, priority areas were identified based on ES capacity (EUC) and ES demanded capacity (EUD), without considering flow zones. The next experiments accounted for flow zones in two different ways, either using NUTS-regions as a uniform approximation of flow zones or using ES type specific flow zones. This implementation interacts with the way Zonation prioritizes locations. Zonation operates by first selecting the entire landscape, and iteratively removing cells that contribute least to the total value of the solution (Moilanen et al. 2014). After each cell removal Zonation updates the remaining cell values, accounting for what has been lost and what remains. In the EU experiments EU-wide ES distributions are used, implying that the value of an ES supplied to city A increases with the removal of a cell supplying the same ES to city B. In contrast, when accounting for flow zones, the EU-wide distribution is broken into many independent flow zones (treated as independent features), and representation of ESs in one flow zone cannot be replaced by representation in another.

The NUTS experiments used EU NUTS-regions as the flow zones of all ESs, using NUTS2-regions for Belgium, Germany and the Netherlands and NUTS3-regions for other Member States (van Berkel and Verburg, 2011). The NUTS-regions consist of aggregates of 1x1km cell and range in area from 12.8 km^2 to 105869 km^2 with a median size of 3614 km^2. One experiment considered ES capacity maps (NUTSC) and another demanded capacity maps (NUTSD). The FLOW experiments explicitly incorporated ES specific flow zones considering ES capacity maps (FLOWC) and ES demanded capacity maps (FLOWD). This approach aims to maintain all ESs distributed across flow zones by accounting for the flow characteristics of each ES. Zonation settings are provided in S17.

Table 4.2: Summary of the six experiments used to identify priority areas, distinguished by experiments with and without accounting for ecosystem service (ES) demand (y-axis) and distinguished by experiments (x-axis) with uniform NUTS-regions (NUTS), ES specific flow zones (FLOW) and without flow zones (EU).

	EU	NUTSx	FLOW
ES capacity	EUC	NUTSC	FLOWC
ES capacity demanded	EUD	NUTSD	FLOWD

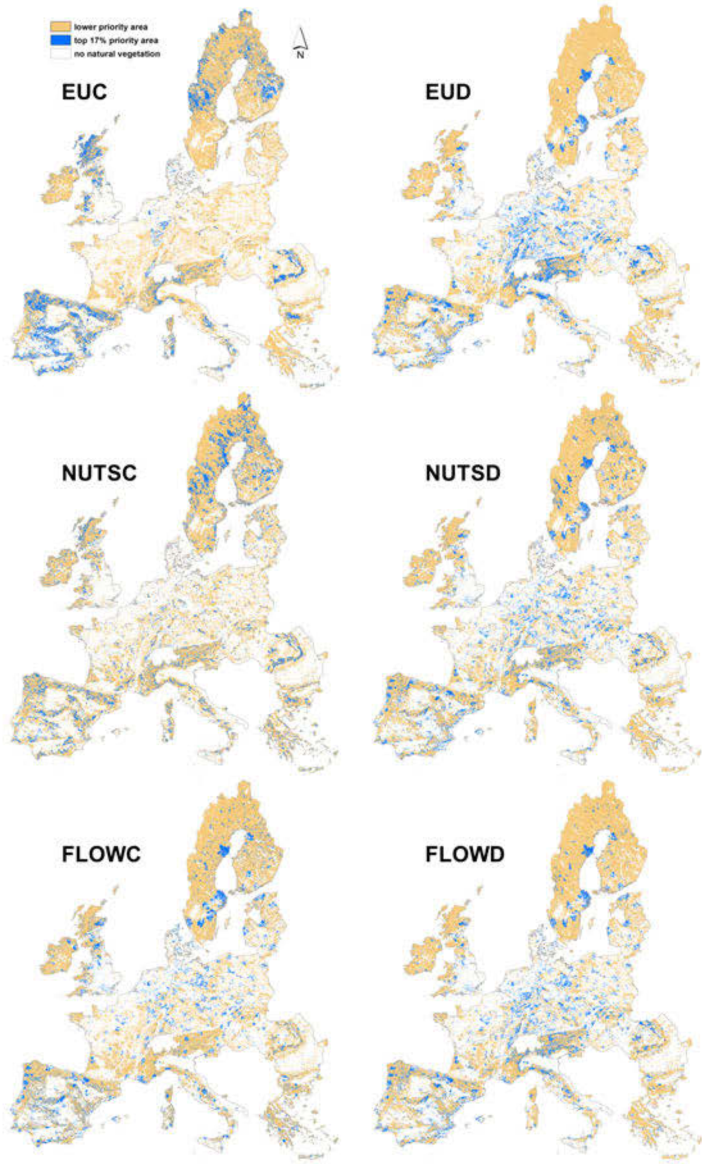
4.2.3 Comparing the experiments

We compared prioritization experiments in three ways. First, we assessed the effect of accounting for demand by comparing the location of priority areas and the level of ES demanded capacity maintained (EUC vs EUD). Second, we tested the effect of accounting for flow zone by comparing the location of priority areas and the level of ES demanded capacity maintained in the EU experiments with NUTS and FLOW experiments. Last, we tested whether only accounting for the ES flow zone (FLOWC & NUTSC) can be used as a proxy for ES demand. Demand maps for many ESs are non-existent at the European scale (Maes et al., 2016) whereas ES capacity maps for more ESs are available, thus making such a simplification an attractive option.

We compared the degree of overlap in priority areas between experiments. The overlap between experiments was calculated for the top 1%, 2%, 5%, 10%, 17% and 25% of the cells ranked as priority area, by calculating the percentage of identical cells in both sets. The 17% area corresponded to the global Aichi target for protected areas. Most results are presented for the top 17%.

The level of ES maintained, always calculated on ES capacity maintained, within the priority areas was calculated at EU and flow zone level. At the EU level we calculated the level of ES maintained with increasing percentage of land retained as priority area. To assess the level of ES maintained per flow zone we calculated two metrics. First, we calculated the percentage of prioritized cells within and outside a flow zone. Priority areas outside flow zones do not have an ES demand. Second, we calculated the percentage of ESs maintained per flow zone within the priority areas. Accounting for flow zones should results in a more even distribution of priority areas over flow zones.

Figure 4.2: Location of 17% of priority areas that maintain the highest level of ecosystem services (ESs) in the EU based on 6 tests related to ES capacity, ES capacity demanded, and ES flow zone (an area with a unique demand-supply combinanation dependent on the ES flow). Tests are defined in section 4.2.2. Only areas containing natural vegetation were considered in the prioritization analysis. Croatia, Malta, and Cyprus are excluded because only partial information on ESs was available for these countries. The percent overlap between priority areas per experiment is in S18.



4.3 Results

4.3.1 Accounting for ES demand

Accounting for ES demand resulted in a clear shift in priority areas in the EU (Figure 4.2). Priority areas for ES capacity (EUC) were predominantly located in remote areas in Europe such as northern Fennoscandia, Western Scotland and the Carpathians, as well as the Iberian peninsula. Accounting for ES demand (EUD) shifted priority areas towards central European countries and towards natural vegetation in the proximity of cities, which is visually apparent for Sweden, Finland and Scotland. The overlap in priority areas between the experiments was low: for the top 1% priority areas only 7.44% is identical, increasing to 37.1%, for the top 25% priority areas (S18).

Accounting for ES demand resulted in a clear increase in the level of ESs maintained in priority areas (Figure 4.3, EUC vs EUD). For all ESs except carbon sequestration the level of ESs maintained was large after accounting for demand. In part this difference was to be expected, as we measured ESs maintained as the demanded capacity maintained, but the magnitude of the difference indicated that most priority areas based on ES capacity are located in areas with no to low demand. The individual ESs responded differently to accounting for ES demand, being most pronounced for local flow ESs. The EUC did not capture much of the demanded capacity of local flow ESs, while the demand experiments captured large fractions of the demanded capacity in a small fraction of the land. Accounting for demand came with a small efficiency loss for carbon sequestration, but the loss was relatively small compared to the gains for other ESs. In other words, our results do not demonstrate a trade-off between global and more local scale ESs when accounting for demand.

The performance of the EUC experiment was especially low for the local flow ESs, because these ESs are very location specific. Air quality regulation is only provided near emission sources. As prioritization based on ES capacity did not consider that only ecosystems close to emission sources contribute to fulfilling ES demand identified priority areas were often outside actual flow zones. For air quality EUC allocated 90% of the top 17% priority areas outside flow zones (Figure 4.4).

4.3.2 Accounting for flow zone

Accounting for the flow zone of ESs was done through the NUTS and FLOW experiments. For the capacity experiments, this resulted in more evenly spread allocation of top priority areas (Figure 4.2 EUC vs NUTSC vs FLOWC). The degree of overlap in top-priority areas between EUC and FLOWC ranged from 16.64%, for the top 1% priority areas, to 50.77%, for the top 25% priority areas (S18). The demand experiments were much more alike in terms of spatial overlap (S18), because accounting for demand already reduced the solution space significantly.

The effect of accounting for ES flow zones varies across ESs. Compared to EUC, the NUTSC experiment has little effect on the demanded capacity retained for most ES, and it performs poorly especially for pollination. For air quality regulation and urban leisure, the FLOWC experiment strongly increases the level of ES maintained compared to the NUTSC and EUC experiments. Accounting for flow zones hardly decreases the level of ES maintained at the EU level for the demand experiments (Figure 4.3).

4

Figure 4.3: The level of ecosystem services (ESs) maintained (percent capacity demanded) in the EU relative to the percentage of conservation-priority areas identified based on 6 tests of ecosystem service capacity, demand and flow zone (an area with a unique demand-supply combination dependent on the ES flow). Different degrees of concavity in the curves result from different size distributions of ESs across the landscape and from the fact that all prioritzations are based on the distribution of five ESs but results are presented per service.

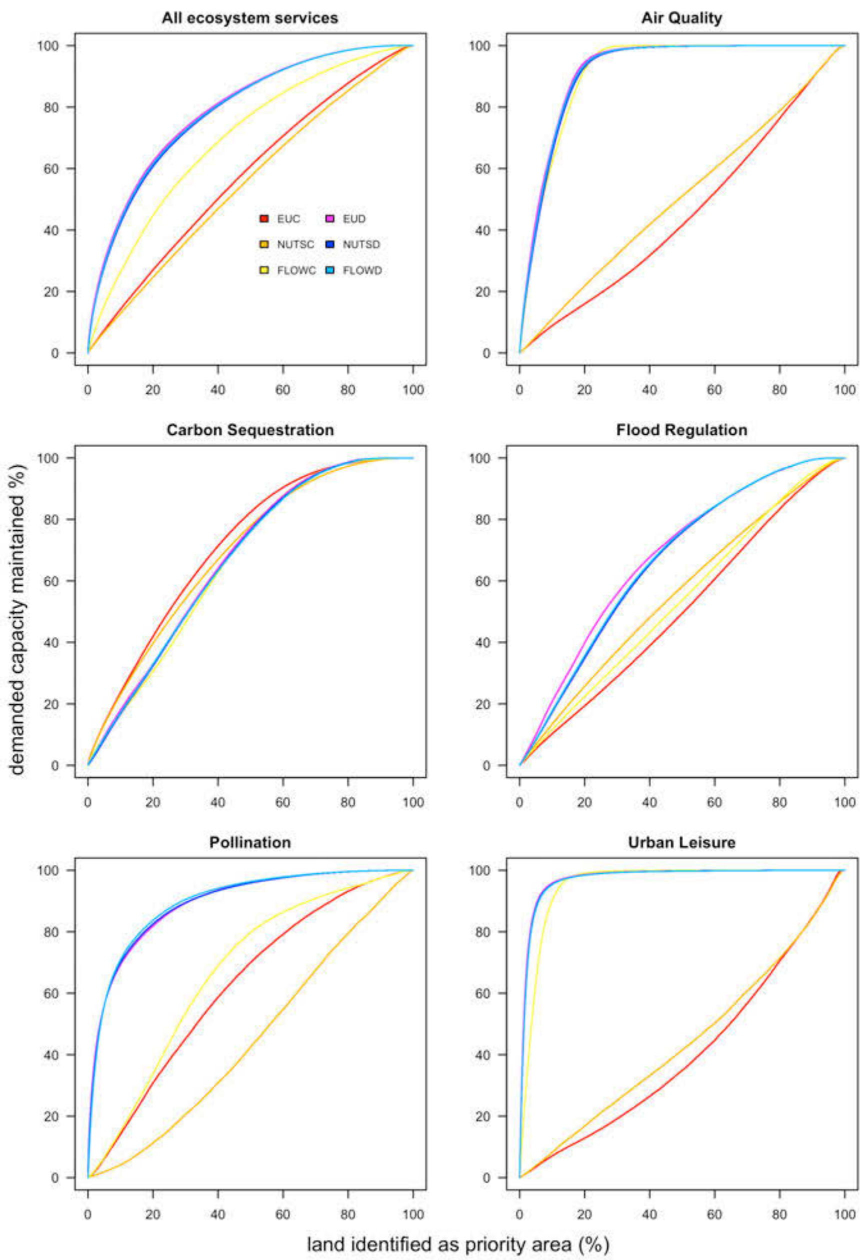
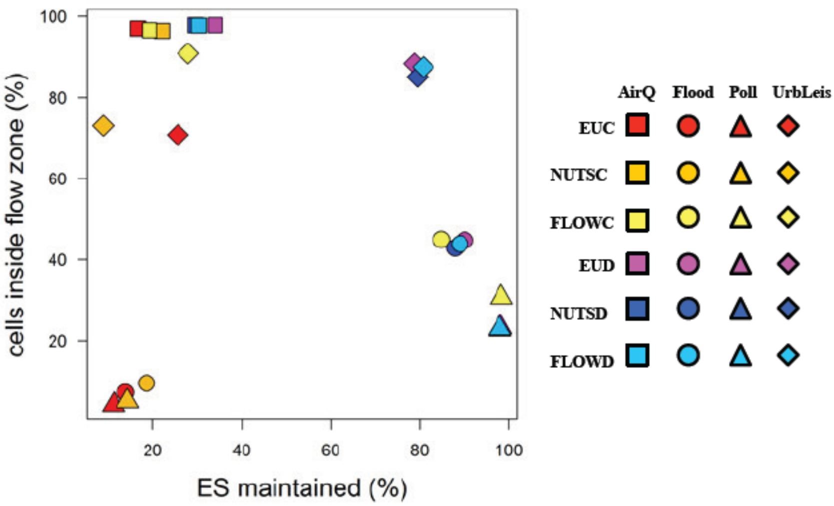


Figure 4.4: Trade-off between the level of ecosystem services (ESs) maintained and the percentage of priority areas within a flow zone. Results are depicted for the 17% of priority areas that maintain the highest level of ESs (airq, air quality; flood, flood control; poll, pollination; urbleis, urban leisure). The percentage of cells within flow zones provides an indication of whether priority areas are located in areas where there is an ES demand at present.



4.3.3 ESs maintained per flow zone

Both accounting for ES demand and flow zones affected the distribution of priority areas across flow zones (Figure 4.4) and the level of ES maintained per flow zone (Figure 4.5). Results differed per ES. We will first discuss air quality and urban leisure. For air quality and urban leisure, the EUC and NUTSC performed similarly having both a low level of ES maintained and a low percentage of priority areas within flow zones. The other four experiments showed high level of ES maintained and a higher percentage of cells within flow zones. A similar pattern was observed for the distribution of ES across flow zones (Figure 4.4). At 17% priority areas there is no effect of accounting for demand on the distribution of ES across flow zones because 100% of the ES is already maintained. In the EUC experiment 26.6% of the flow zones have no ES maintained. In the FLOWC and demand experiments all flow zones have at least some ES maintained. At 5% or 10% priority areas there was a redistribution of ES maintained across flow zones (S18).

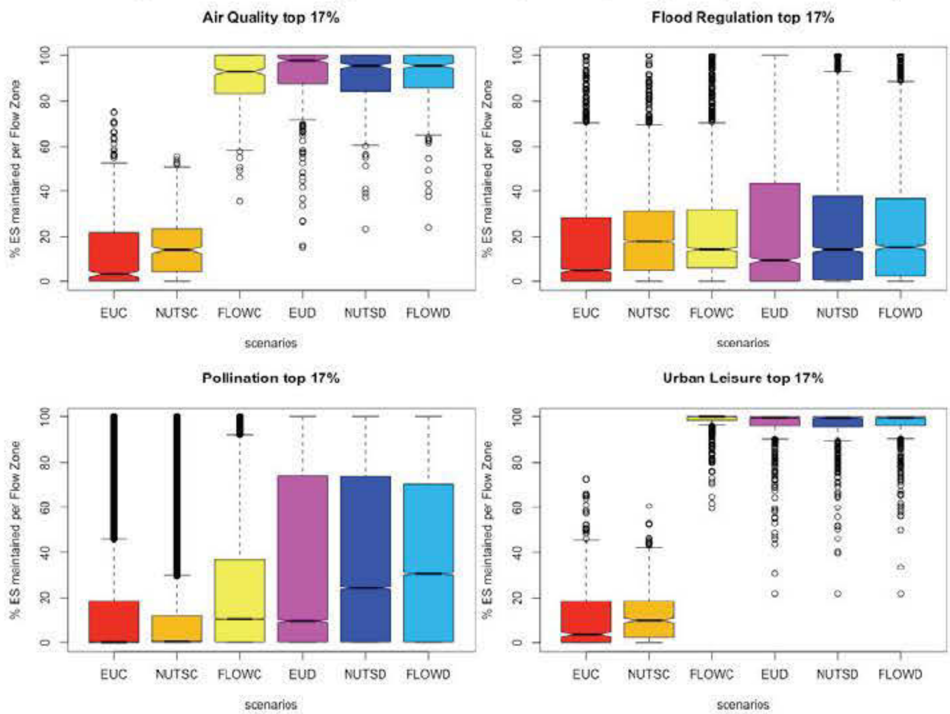
For flood regulation and pollination there was a clear difference between the capacity and demand experiments. All experiments had a high percentage of priority areas within flow zones, mainly because the flow zones cover almost all of Europe. FLOWC performed similar to the other capacity experiments although for pollination more cells are maintained within flow zones. Accounting for ES demand clearly increased the level of ES maintained, but there was no clear difference between the demand experiments (Figure 4.5). Of the demand experiments FLOWD had a higher median combined with the lowest variation of the demand experiments, meaning that priority areas were more evenly distributed over flow zones. Moreover, accounting for the flow zone (FLOWD vs. EUD) reduced the amount of flow zones with no ES maintained for flood regulation (8.9%) and pollination (14.3%) (S18).

4.3.4 The flow zone as a proxy for ES demand

We evaluated the NUTSC and FLOWC experiments for the possibility to use flow zones in combination with ES capacity maps as a proxy for ES demand. Spatially, the NUTSC experiment caused a larger spread of priority areas compared to EUC (Figure 4.2), but this attempt to better distribute priority areas among areas of demand also resulted in priority areas in regions with little demanded capacity (Figure 4.3). The FLOWC experiment did show close resemblance with the demand experiments. The overlap in priority areas between all demand experiments and the FLOWC experiment ranged from 22.3% for the 1% priority areas, to 62.3% for the 25% priority areas.

The FLOWC experiment increased the level of ES maintained for air quality and urban leisure resulting in a similar performance compared to the demand experiments (Figure 4.3). The FLOWC experiment resulted in a smaller improvement for the other ESs. The NUTS experiment does not increase the ES maintained per flow zone compared to the EUC experiment for all ESs.

Figure 4.5: Percentage of ecosystem services (ESs) maintained per flow zone for the 4 ecosystem services for each test (x-axis; test defined in section 4.2.2). A small bar (colored area) and a high median (black lines) indicate a relatively even distribution of the level of ESs maintained per flow zone. Dashed lines indicate the value at 1.5 times the interquartile range (i.e., the difference between the minimum and maximum value of the bar). Values outside this range are often outliers (circles). Circles indicate flow zones that have far higher or far lower levels of ESs maintained than is expected based on the spread of the data. The results shown are for the top 17% priority areas that maintain the highest level of ESs. Figures for different percentages of priority areas are in S18.



4.4 Discussion

4.4.1 Reflection on results

This study aimed to quantify the importance of accounting for the demand and flow zone of ESs in identifying priority areas. Our results demonstrated that only accounting for ES capacity data is likely to result in priority areas high in ecosystem functioning but that do not actually provide ESs to society. In particular for local flow ESs it is important to consider the fraction of the capacity that fulfills a demand, rather than capacity per se. Our findings are in line with research on ES conservation in Canada where priority areas for ES capacity maintained between 20% and 50% of the proportion of ES demanded capacity for local flow ES (Cimon-Morin et al., 2014).

As benefits from local and regional ESs need to be distributed across the EU, we assessed the effect of accounting for flow zones. Our study moves beyond the traditional way of evaluating priority areas for efficiency at the study scale, by including metrics on the spatial distribution of ESs and priority areas.

We find that accounting for flow zones changed the location of priority areas. The redistribution of priority areas across flow zones did not affect the level of ES maintained, but did result in a reduction of flow zones without any ES maintained. Moreover, accounting for flow zones resulted in the level of ES maintained per flow zone to become more alike. This effect was strongest for small priority networks for air quality and urban leisure and for large priority networks for flood regulation and pollination. In general, accounting for the flow zone resulted in a more even distribution of ES maintained across the EU without a clear decrease in the total level of ES maintained. Nevertheless, the effect of accounting for flow zones is small and the effect differs per ES. Previous research has mostly ignored flow zones in prioritization (Casalegno et al., 2014; Cimon-Morin et al., 2014). Chan et al. (2006) did identify flow zones per ES, but did not test the effect of accounting for flow zones on ES prioritization. We extended the capacity of Zonation beyond the possibility to account for uniform administrative units (Moilanen and Arponen, 2011; Pouzols et al., 2014). In our approach it is possible to combine ES specific flow zones within spatial prioritization.

Although ES demand is more frequently mapped nowadays (Wolff et al. 2015), maps for ES demand and ES flow are not commonly available. We show that for prioritization ES specific flow zones (FLOWC) can be used as a proxy for ES demand, particularly for local flow ESs, in the absence of demand data (Figure 4.3). Using administrative units as a generic proxy for demand (NUTSC) did not provide equally good results in terms of ES demanded capacity maintained (Figure 4.3).

4.4.2 Approach

Our study considered five ES for which both ES capacity and ES demand data were available. The results are partly driven by this selection of ESs. ES demand is especially high close to cities and the resulting priority network is strongly driven by demand for air quality, urban leisure and to a lesser extent flood regulation. To fully assess the efficiency of the prioritized areas the over- and under-supply of ESs per flow zone or for the EU would be interesting to consider. However, such analysis was not possible as ES demand and ES capacity data were not measured in the same units, except for air quality regulation. The approach used here to calculate demanded capacity is straightforward and easy to implement for all ES flows, as long as there is an estimate of demand that is spatially explicit.

Flow zones are not commonly used in prioritization studies. Delineation of flow zones was, except for pollination, based on previous studies and existing spatial planning units. Previously Chan et al. (2006) identified flow zones using similar delineation methods including catchments (flood control) and city surroundings (recreation). Chan et al. (2006) did not identify flow zones for pollination because of uncertainties in foraging distances of pollinators.

More detailed delineation of flow zones and accounting for the uncertainty in current delineations could further improve these assessments.

A full ES prioritization needs to incorporate additional variables such as management costs, human based alternatives for ESs and threats to ES supply (Luck et al., 2012). In our study we used area as a costs measure, as has been done by others before (Casalegno et al., 2014; Pouzols et al., 2014). Non-uniform costs can be assessed through costs for land acquisition and land management (Naidoo et al., 2006; Remme and Schröter, 2016) or through costs of foregone production (Schröter et al., 2014b). Land costs could be approximated using land prices or, if not available, using population density and GDP as a proxy. Land prices are likely higher around cities and could therefore conflict with priority areas after accounting for ES demand. Our results indicate that, in spite of the potential costs, priority areas for ESs are likely to remain close to cities given the high ES demand in the cities' neighborhood. Therefore, we consider our result that accounting for demand causes large shifts in priority areas to be robust.

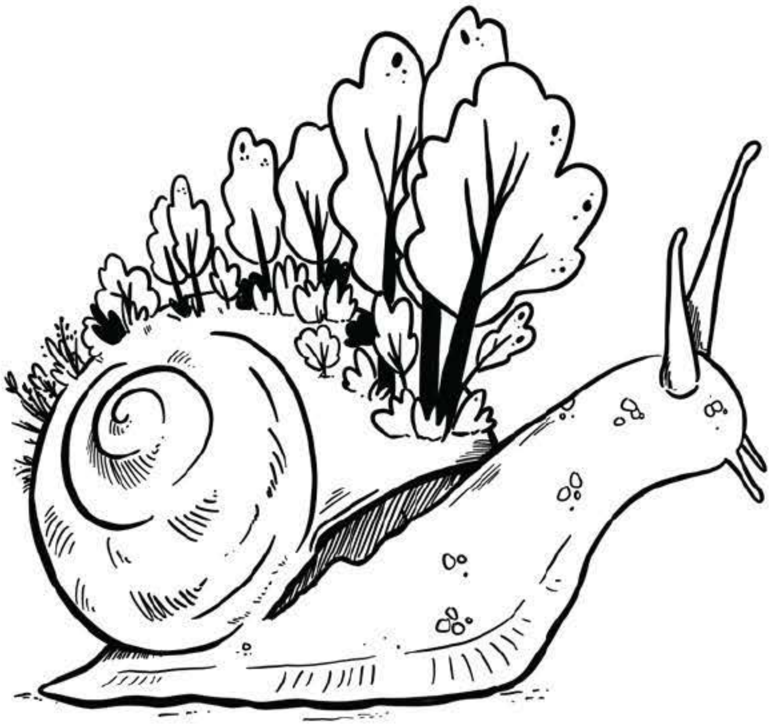
In our prioritization approach we did not account for land use change or other threats to ESs. Land use change can affect both ES capacity and ES demand (Stürck et al., 2015b). For example, increases of biofuel crops, such as oil seed rape, might increase the pollination dependency of current croplands. Land use change might results in threats to ESs (urban expansion) but could also provide opportunities for restoration of ESs following land abandonment.

There are important differences between Zonation, as used in our study, and other spatial prioritization software such as Marxan (Possingham et al., 2000). Zonation generates a priority ranking through the study area instead of trying to achieve a target-based solution (Lehtomäki and Moilanen, 2013). Zonation is most useful when individual targets cannot be easily determined (Lehtomäki and Moilanen, 2013) as is the case for most ESs (Remme and Schröter, 2016). Accounting for flow zones is possible in target based planning software such as Marxan. In a study in California, Chan et al. (2006) accounted for flow zones of ESs by assigning unique conservation targets to different zones using Marxan.

4.4.3 Policy implications

Our outcomes have important consequences for policies aimed at protecting biodiversity and ESs. Accounting for the flow zone of ESs resulted in smaller and more scattered priority areas. For biodiversity larger clustered protected areas are preferred because of management efficiencies and species habitat requirements (van Teeffelen et al., 2006). Moreover, coordinated identification of priority areas results in more efficient solutions for biodiversity conservation (Pouzols et al., 2014; Kukkala et al., 2016). Priority areas for biodiversity also need to balance between efficient biodiversity protection and the distribution of biodiversity conservation across a region. The most efficient protection network could simultaneously create a politically and biologically undesirable outcome if it does not result in maintaining well-functioning natural systems across an area (Moilanen and Arponen, 2011). The need to protect biodiversity at a global scale and ESs at smaller scales therefore does not have to be at odds.

Most importantly, our results clearly show that ignoring ES demand leads to the identification of priority areas in remote regions where benefits to society are small. Incorporating ESs in conservation planning should therefore always account for ES demand to avoid less efficient solutions.



5. Identifying spatial priorities for ecosystem services across Europe under land use change

Abstract

Policy objectives to maintain ecosystem services are increasingly set. Methods to identify priority areas for ecosystem services can assist in the implementation of such policy objectives. While land use change is an important driver of changes in ecosystem services over time, most prioritization studies do not account for land use change or only assess negative effects. We assessed the effect of land use change on ecosystem services in Europe for a 40-year period and the subsequent consequences for identifying priority areas.

We quantified five services under current and future land use. For both time frames all sites were ranked based on their service provision using Zonation. To assess the sensitivity of the prioritization to land use change we compared the location of priority areas and the level of ecosystem services within priority areas in the two time frames.

Land use change shifts the location of priority areas. Overlap in priority areas over time ranges from 34.8% overlap for the top 1% priority areas to 75.4% overlap for the top 25% priority areas. Moreover, land use change affects the availability of ecosystem services in top priority areas: Compared to current top priority areas, future top ranked priority areas have lower pollination and carbon sequestration capacity. Capacity of erosion control and flood control are stable over time and nature-based tourism increases.

Shifts in priority areas are driven not only by local land use change, but also by land use change in the wider landscape, through connectivity effects and shifts in the relative importance of sites. The real management challenge lies in maintaining ecosystem services within landscapes where production and conservation objectives need to be reconciled and priority areas are affected by both local and landscape wide changes in land use. Moreover, we show that land use change has both local positive and negative effects on ecosystem service priorities, indicating that prioritization studies should not solely incorporate negative effects of land use change.

This chapter is published as:
Verhagen, W., van Teeffelen, A.J.A. and Verburg, P.H. (2018), Shifting spatial priorities for ecosystem services following land use change. Ecological Indicators, 89: 397-410. doi: 10.1016/j.ecolind.2018.01.019

5.1 Introduction

Multiple competing demands for land exist, ranging from the production of agricultural and forestry products to the need for recreational spaces and a healthy living environment. Ecosystem Services (ESs) are affected by different facets of land use change including land cover conversion, (de)intensification of land management and changing the spatial arrangement of land cover types (Seppelt et al., 2016). In order to maintain ESs, policies aim to protect land with high values for ESs alongside biodiversity (Convention on Biological Diversity, 2010). To assist implementation of these policies researchers have developed approaches to prioritize areas for services (Cimon-Morin et al., 2013; Remme and Schröter, 2016; Verhagen et al., 2017). Prioritization analysis ranks landscape units with respect to the occurrence of multiple ESs and is often part of a wider approach for systematic conservation planning, aimed at identifying the most cost-effective protected area network (Moilanen et al., 2009). Priority areas are the highest ranked areas, often identified based on a predefined threshold, together providing the highest amount of ESs. Most prioritization analyses for ESs are based solely on the current state of land use and ESs (Luck et al., 2012). Such analyses, therefore, do not address the maintenance of ESs over time and do not support the development of management strategies to alleviate the impacts of land use change.

Within a conservation planning framework land use change is mostly considered as a threat to ESs (Luck et al., 2012; Cimon-Morin et al., 2016). Incorporating multiple ESs alongside agricultural objectives in land use allocation generates higher benefits from land use as compared to land use allocation based solely on agricultural objectives (Bateman et al., 2013). Land use change can however have both positive and negative effects on ESs. Studies have indicated that for the European Union over time ES capacity increases for some services while decreasing for others, and is driven by changes in climate but especially by changes in land use (Schröter et al., 2005; Polce et al., 2016). At the local and regional level the effect of land use change varied from strongly positive to strongly negative, irrespective of the general trend in ES capacity at the EU level (Schröter et al., 2005; Metzger et al., 2006; Stürck et al., 2015b; Polce et al., 2016). The local effects of land use change on ES capacity depend on the type of land use change, the service considered and the local biophysical context. Prioritization studies should thus account for both positive and negative effects of land use change on ESs.

A number of previous studies have linked land use change to ES models and prioritization techniques. Some studies directly link land use change to the prioritization analysis, without quantifying ESs under future conditions. One example of this are studies that exclude areas from the prioritization network based on projected land use change or development (Troy and Wilson, 2006; Cimon-Morin et al., 2016). Another example thereof are studies that assign positive and negative weights in the prioritization analysis to areas with projected land use change (Phua and Minowa, 2005; Luck et al., 2009; Wendland et al., 2010; Nagendra et al., 2013; Pouzols et al., 2014). The main limitation of these approaches is that the effect of land use change on ESs is often assumed uniform irrespective of the local context. Other studies use the assessment of ESs as an intermediate step in the prioritization analysis. Quantifying ESs under both current and future land use simulations can account for non-uniform responses of ESs to land use change based on local conditions. This type of approach has been mostly used to study the effect of a single land use change process, such as deforestation and urbanization, on future ESs and prioritization (Reyers et al., 2009; Venter et al., 2009). Recently, Fan et al., (2016) prioritized areas in a watershed in Japan for multiple ESs and multiple land use change processes using this approach.

The provision of ESs can depend on local close by or global far-off ecosystems depending on the characteristics of ES flow (Fisher et al., 2009). ES flows are the connections between areas with ES capacity and those with ES demand. Not accounting for ES demand results in the identification of priority areas with high ES capacity but possibly low ES provision to beneficiaries (Cimon-Morin et al., 2014; Verhagen et al., 2017) but data on ES demand is not commonly available (Wolff et al., 2015; Maes et al., 2016). This can be corrected for by incorporating flow zones as a proxy for ES demand (Verhagen et al., 2017). Flow zones constrain the prioritization by only considering ES capacity in areas that have a potential demand for that service. Air quality regulation is only a service to humans in populated areas and flood regulation only provides a service to humans in watersheds with human-use. Furthermore, an area can consist of multiple flow zones where each zone is limited by the extent of the ES flow, i.e. multiple watersheds. For local and regional services ES capacity can only contribute to a demand if the ES capacity is maintained within the flow zone. In other words, protecting flood regulation capacity in a watershed in Finland does not provide flood regulation in a watershed in Germany. Depending on the service considered, local losses of the service following land use change cannot be offset by distant gains in that service (Verhagen et al., 2017). Given that many ESs are supplied at sub-global scale it is of particular importance to analyze the effects of land use change on ES prioritization at both global and sub-global scale.

The aim of our study is to assess the effect of land use change on the priority ranking of areas for ESs. Here we define priority areas as areas above a certain rank (e.g. top 10%), which together provide the highest level of ESs. Using the European Union (EU) as our study area, we specifically ask: 1) to what extent does projected land use change shift the location of priority areas and does it affect the level of ESs maintained within these areas; and 2) how are such shifts spatially distributed across regions. Land use change projections in Europe are characterized by a combination of locally intensifying land management and large-scale agricultural land abandonment. These contrasts make Europe a suitable case study to test the interplay between potentially positive and negative impacts of land use change on prioritization for ESs. Our study provides spatially explicit information for managing priority areas by identifying on the one hand current priority areas that might be vulnerable to the negative effects of land use change, and on the other hand areas where favorable land management can provide opportunities for future priority areas for ESs.

5.2 Methodology

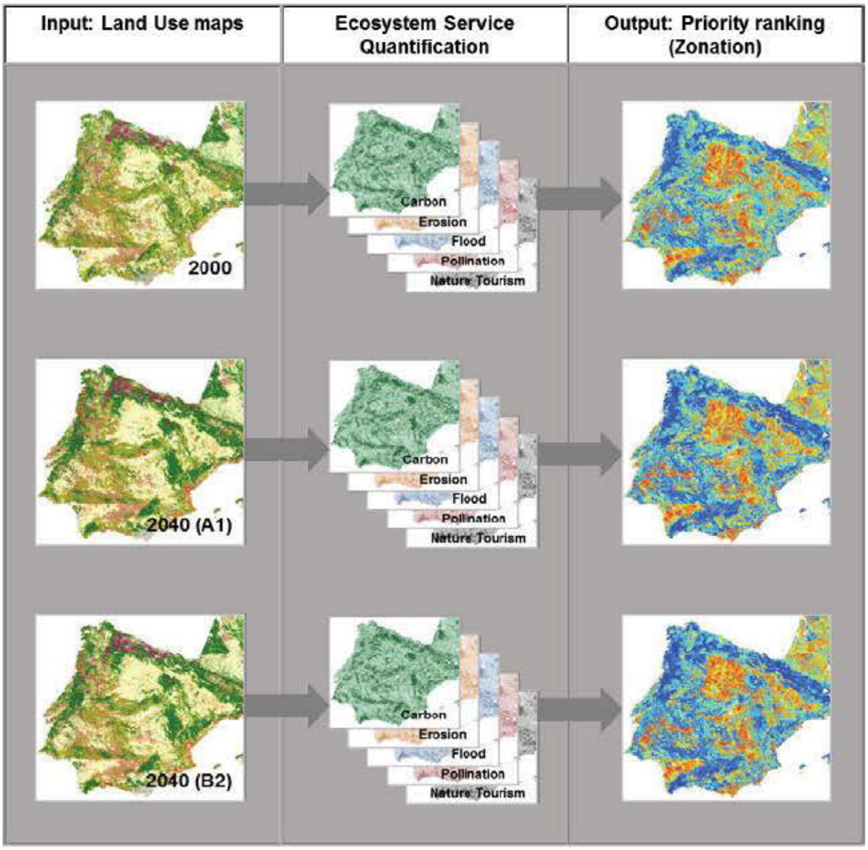
Our analysis consists of three major parts (Figure 5.1): 1) modeling current and future land use (section 2.1); 2) The quantification and mapping of ESs capacity (section 2.2); and 3) the prioritization analysis (section 2.3). In section 2.4 we explain how the results of the different experiments are compared.

5.2.1 Current and future land use in Europe

We used an existing set of EU land use change scenarios that simultaneously project changes in land cover and land management between the years 2000 and 2040 (Stürck et al., 2015a). We briefly describe the main features of the land use simulations, and refer to Stürck et al. (2015a) for details.

We used a consistent modeling chain linking data on land cover, socio-economic drivers, land use change at an appropriate level of detail required to predict changes in related ESs (Stürck et al., 2015a; Lotze-Campen et al., 2017). Data on land cover and land management intensity in 25 EU member states (excluding Croatia, Cyprus and Malta) was compiled for the years 2000 ("current", given a lack of consistent data for all variables of more recent date) and for two land use scenarios in 2040 ("future"). For the year 2000, CORINE land cover data were thematically aggregated to 16 land cover class at a 1km resolution (Table 5.1). Land management intensity was assessed for croplands, grasslands and forests. Management intensity on croplands was determined using nitrogen application (Overmars et al., 2014), on grasslands using stocking density (Temme and Verburg, 2011) and in forests using wood removal (Verkerk et al., 2014). Land management intensity was provided as an output at regional (NUTS2) level and disaggregated to 1km^2 cells (Stürck et al., 2015a).

Figure 5.1: Schematic overview of the analysis. We have a total of three land use maps which are used as an input to calculate the capacity of five ecosystem services. These ecosystem service maps are the input for the prioritization analysis which results in a full priority ranking of the landscape per experiment. For the prioritization analysis in Zonation the ecosystem service input maps are split by flow zone, meaning that erosion, flood control and pollination each have multiple input maps. The maps presented here are only used for display purposes and do not represent the input or output maps of the analysis.



For 2040, land use projections were derived from Dyna-CLUE model simulations (Verburg and Overmars, 2009). The Dyna-CLUE model combines top-down and bottom-up drivers to allocate land use change using outputs of global scale macro-economic models as an input. We used Corine Land Cover for the year 2000 to allow for a start-up period to capture the temporal path-dependence of land abandonment and regrowth more accurately (Verburg & Overmars, 2009). A detailed description of the land allocation algorithm in Dyna-CLUE can be found in Verburg and Overmars (2009). Scenarios for future land use change followed the IPCC SRES storylines (A1, A2, B1 and B2) but were adjusted to fit the European context (Nakicenovic et al., 2000; Paterson et al., 2012). In this study we used the two most diverging scenarios in terms of dynamics of agricultural area, namely A1 (high agricultural abandonment) and B2 (relatively low agricultural abandonment). In Europe land abandonment is one of the dominant land use change processes impacting a wide set of environmental indicators (van der

Zanden et al., 2017). The A1 Scenario ("Libertarian Europe) is characterized by a globalizing world with strong economic growth, moderate population growth and growing demand for food and feed. Environmental policies are not enforced. The B2 scenario ("European Localism") is characterized by a fragmented world with moderate economic growth, moderate population growth and moderate growth of the demand for food and feed. Environmental policies are in place but implemented at a regional level. The current land use map, the A1 future land use map and the B2 future land use map, all at 1 km resolution, were used as input to the ES models.

We quantified and mapped four regulating (carbon sequestration, erosion control, flood regulation and pollination) and one cultural ES (nature-based tourism) under current and future land use. Regulating and cultural ESs are generally thought of as conservation compatible, i.e. benefits obtained from these services dot not negatively impact biodiversity conservation (Chan et al., 2006; Schröter et al., 2014b). Each ES was quantified for the year 2000 and for the year 2040 (scenarios A1 & B2) resulting in a total of five ES maps per experiment. We next explain the quantification of ESs and how land use change affects ESs. A full description of the quantification of ESs over time is provided in Stürck and Verburg (2016).

Table 5.1: Overview of land cover classes for the years 2000 and 2040. The land cover classes are based on the land cover classes in the DynaCLUE model (Verburg and Overmars, 2009; Stürck et al., 2015a). Information on land management intensity is not incorporated.

Land cover classes
Built-up area
Arable land
Pasture
Semi-(natural) vegetation
Inland wetlands
Glaciers and snow
Irrigated arable land
Recently abandoned arable land (only in future predictions)
Permanent crops
Forest
Sparsely vegetated areas
Beaches, dunes and sands
Salines
Water and coastal flats
Heather and moorlands
Recently abandoned pasture (only in future predictions)

Carbon sequestration

Carbon sequestration (Mg C km^2 yr^{-1}) is an important global ES provided by soils and vegetation. Land use can result in both net carbon emissions and net carbon uptake. Total carbon sequestration is calculated as the sum of 1) changes in soil carbon sequestration under all land use measured using a dynamic bookkeeping approach (Schulp et al., 2008), and 2) changes in forest biomass sequestration under changing forest management and aging forests (Verkerk et al., 2014). Yearly tree biomass increment depends on the age class of the forest and on the total wood volume (Schelhaas et al., 2007). Management negatively affects carbon sequestration due to wood removal and in subsequent years results in higher carbon sequestration rates due to younger forest stands resulting in higher yearly wood increment (Verkerk et al., 2014). Changes in sequestration by soils thus only depend on land cover whereas changes in sequestration by biomass depend on both changes in land cover and land management. Initial carbon stocks and carbon sequestration rates are quantified using emission factors depending on country and land cover type. In order to solely focus on ESs and not disservices, all negative carbon sequestration values, i.e. emissions, were set to zero. Our carbon sequestration model only accounted for annual carbon sequestration rates and did not account for changes in stored carbon stocks in soil and vegetation.

Erosion control

Erosion control (t/ha/year) is defined here as the local protection of soils against erosion. We used the erosion control model described in Tucker et al. (2013) which quantified soil erosion based on the USLE equation (Wischmeier and Smith, 1978). The USLE equation combines data on climate, soil type, slope length and land cover type to calculate erosion risk. To distill the contribution of the land cover type to alleviating soil erosion risk we subtracted the actual soil erosion (i.e. the soil erosion with land cover) from the potential soil erosion (i.e. the soil erosion without land cover) (Luck et al., 2012; Verhagen et al., 2016). Erosion control is also influenced by land management on arable land such as contour farming or the presence of stone walls (Panagos et al., 2015) but not by the intensity of fertilizer use. Therefore, we did not account for management intensity and erosion control is only affected by changes in land cover type.

Flood Regulation

Flood regulation is here defined as the interception and infiltration of runoff towards streams due to land cover. Flood regulation was quantified based on a look up table approach incorporating precipitation regime, catchment type (e.g. mountainous), location of land cover in the catchment and water holding capacity (Stürck et al., 2014). The water holding capacity is a factor of both the land cover type and management intensity of all land cover types. Flood regulation is quantified as a dimensionless index ranging from 0-1. Changes in land cover and management intensity directly affect flood regulation. All other parameters were assumed constant over time (Stürck et al., 2015b).

Nature-based tourism

Nature-based tourism denotes the capacity and attractiveness of ecosystems to support nature tourism. Nature-based tourism was quantified based on the method of Van Berkel and Verburg (2011) including as indicators elevation difference and the presence of lakes, rivers and protected areas (van Berkel and Verburg, 2011). Nature-based tourism additionally depends on the surrounding landscape which was classified into dominant urban, agricultural or forest landscape and a mixed or mosaic landscape (van Berkel and Verburg, 2011). We used an updated version of this method by van der Zanden et al. (2017). This updated version excluded areas with EU's high nature value farmland policies as an indicator because the future of this subsidy scheme is already included in the A1 and B2 land use scenarios. Over time nature-based tourism changes due to changes in land cover type, locally, or through changes in surrounding land cover. The quantification solely focused on the ecosystems' capacity to support nature-based tourism and does not account for use or demand through proxies like accessibility.

Pollination

Pollination, here limited to pollination by wild bees, is an important local flow ES for the production of certain food and biofuel crops. We mapped pollination flows between natural vegetation and croplands based on a combination of two existing studies (Zulian et al., 2013; Serna-Chavez et al., 2014). Pollination flow is sometimes assigned to croplands but from a conservation perspective pollination capacity needs to be assigned to natural vegetation, i.e. the main wild bee pollinators' habitat. Therefore we made some adjustments to the approach of Serna-Chavez et al. (2014). All land cover types were assigned a nesting suitability score based on a look-up table from Zulian et al. (2013), generally assigning high values to semi-natural vegetation and forest edges. However, pollination requires a service flow from habitat areas to croplands. Field studies across Europe showed that wild bee diversity on croplands was separately affected by the proximity of habitat areas and the management intensity of the cropland (Hendrickx et al., 2007). We categorized management intensity of cropland into three classes based on fertilizer application (low: <50 kg/ha, medium; 50 -150 kg/ha and high: >150 kg/ha) (Stürck et al., 2015). To account for the proximity of cropland to natural vegetation we calculated the area of cropland (0-100%) in the eight cells directly surrounding each habitat site. In this neighborhood each cropland contributed proportionally, depending on the management intensity to account for the negative impacts of intensive management on pollinators. High intensity croplands have been found to host approximately only half the wild bee diversity of lowest intensity croplands (Hendrickx et al., 2007). Wild bees are important pollinators for many crops and studies on apple orchards highlight that increased species richness of wild bees resulted in increased fruit set (Garibaldi et al., 2013; Mallinger and Gratton, 2015). We therefore assumed that low intensity cropland had a maximum 100% area contribution, medium intensity cropland had a reduced 75% area contribution and high intensity cropland had a reduced 50% area contribution. Highest pollination values were thus assigned to cells with forests or semi-natural vegetation surrounded by low intensity croplands. Over time pollination capacity was affected by changes in the local land cover type of the habitat cells and by changes in land cover and management intensity of croplands in the proximity of natural vegetation.

5.2.2 Prioritization approach

We used the software package Zonation v4, a tool developed to identify the most cost-effective network for conservation of biodiversity or other features (Moilanen et al., 2009, 2014). Zonation has been previously used to identify ES priority areas in e.g. England, Japan and the European Union (Casalegno et al., 2014; Fan et al., 2016; Verhagen et al., 2017). Zonation assigns a continuous ranking to all cells in a landscape based on the occurrence of features of interest (here: ESs), iteratively removing the cells with the lowest score for the features (Lehtomäki and Moilanen, 2013). After removing the lowest ranked cells from the prioritization the values of the remaining cells are updated based on the amount of ES in the lost cells relative to the amount of ES within the remaining cells. In other words, for abundant ESs the penalty of losing part of the ES value in the beginning is low but increases when the ES becomes scarcer. Zonation differs from target based land planning approaches that have been used for ES prioritization including Marxan (Chan et al., 2006), Marxan with Zones (Schröter and Remme, 2016) or C-Plan (Cimon-Morin et al., 2016). Zonation is most useful when individual targets per ES cannot be set, an issue for most services (Remme and Schröter, 2016).

We used flow zones as a proxy for ES demand by only identifying priority areas in locations with ES demand (Verhagen et al., 2017). As a pre-processing step for the prioritization analysis the European map per ES is split into individual maps per ES specific flow zone. Zonation treats each flow zone input map as a separate protection features resulting in a better distribution of priority areas across flow zones in Europe with minimum impact on the overall protection of each ES (Verhagen et al., 2017). The identification of flow zones differs per ES and partly depends on the type of ES flow. Nonetheless there is yet no uniform way of identifying flow zones per ES and therefore we chose an approach that is both biophysically relevant but also computationally feasible. Carbon sequestration is a global flow ES and is not spatially restricted, due to the global climate system Therefore, we use the entire EU as flow zone for this service. Nature-based tourism was also considered as a global flow ES, due to the global nature of today's tourism. We define nature-based tourism as nature recreation in larger areas lasting multiple consecutive days. For this type of nature-based tourism there is no absolute limit on the travel time or distance. In contrast, more localized day to day recreation services including urban leisure in green spaces can be restricted to individual city areas (Verhagen et al., 2017). As we restricted ourselves to tourism rather than recreation, we use the entire EU as flow zone for this service. Flood regulation and erosion control are dependent on the movement of water and soil particles between and within catchments. We used 5th order catchments with a minimum area of 2 km^2 to delineate flow zones for these two ESs (EEA 2008). For the prioritization analysis this means that at each cell-removal round (i.e. a step in which cells that contribute least are removed from the set of remaining priority areas) only the values of those cells belonging to the same catchment as where cells were removed from, are updated. Pollination is a very localized ES flow restricted by the flight distance of wild bee species (Ricketts et al., 2008). Following Verhagen et al. (2016a),we split the pollination services map into 10x10km zones but only included zones containing agricultural land. This resulted in approximately 38,000 zones (and equally many input files) for pollination. Flow zones for carbon sequestration, erosion control, flood regulation and nature-based tourism are considered stable over time. For pollination, agricultural expansion and abandonment can result in shifting flow zones over time. To account for this flow zones were identified as locations with agricultural land under current or future land use. Therefore, potential shifts in the location of flow zones for pollination did not affect the results.

We ran a total of three Zonation experiments, one per land use projection (2000, 2040A1, 2040B2), using their respective set of five ES maps as input (Figure 5.1). Every prioritization analysis requires the setting of parameters and here we mention the most important ones. See S20 for all settings. All experiments had equal settings, only the actual input (ES maps) differed per experiment. In Zonation, one needs to define how the individual ES maps are aggregated to calculate a combined ES value per cell. We used Zonation's aggregated benefit function, which sums the ES value of all features per cell thus prioritizing areas that can cost-effectively cover multiple ESs (Lehtomäki and Moilanen, 2013). Each input file needs to be assigned a weight. For each ES the number of input files is equivalent to the number of flow zones. The aggregated weight assigned to each ES is 100. For each service the aggregated weight is equally distributed over the number of flow zones of that service. Finally, Zonation requires the setting of costs for the analysis. Incorporating non-uniform costs results in more cost-efficient protected area networks for biodiversity (Naidoo et al., 2006). Here we however used land as a uniform cost measure because we were only interested in the effect of land use change on ES capacity and it is unknown how conservation costs will change over time. All Zonation analyses were run using the SurfSARA HPC cloud facilities for high performance computing.

5.2.3 Analysis of prioritization results

For each experiment the output comprised a full ranking of the landscape based on the occurrence of ESs at a 1km resolution (Figure 5.1). The following assessments were made: 1) we assessed the degree of congruence in priority rankings between experiments; 2) we determined the type of land use change current and future priority areas were subject to; and 3) we assessed how the level of ESs in priority areas changed.

We assessed the congruence between top ranked areas of each experiment calculated as the percentage of priority areas in experiment A that is also a priority area in experiment B. To assess the sensitivity of the result to the size of the priority area network chosen we analyzed the results for the top 1%, 2%, 5%, 10%, 17% and 25% ranked cells. We included the 17% threshold as it relates to the Aichi target 11 to protect 17% of the land globally for biodiversity and ESs (Convention on Biological Diversity, 2010). For each set of top X%

priority areas we calculated the degree of spatial congruence (i.e. level of agreement in terms of sites included in the top X%) between all combinations of the three experiments. We did not consider changes in the specific ranking of sites *within* each top X%.

Table 5.2: Overview and description of land use changes. Most land use change categories follow the classification from Sturck et al. (2015). For visualization purposes some land use changes are grouped according to the classification in column 3.

Trajectory class	Trajectory Name	Description
Land cover Conversion	Stable	No change in either land cover or land management intensity
	Land abandonment cropland	Change from arable land to natural vegetation or recently abandoned land
	Land abandonment pasture	Change from pasture land to natural vegetation or recently abandoned land
	Crop2pasture	Change from arable land to pasture
	Crop conversion	Change from one category of arable land to a different category
	Recultivation pasture	Change from pasture to arable land
	Natural succession	Change from recent abandoned land or semi natural vegetation to forest land cover
	Recultivation forest	Change from forest to arable land
	Urbanization	Change from any land cover category to urban or peri-urban area
Change in land management	Cropland extensification	Stable arable land category, reduction in fertilizer
	Cropland intensification	Stable arable land category, increase in fertilizer use
	Pasture extensification	Stable pasture land cover, reduction in stocking density
	Pasture intensification	Stable pasture land cover, increase in stocking density
	Forest extensification	Stable forest land cover, reduction in wood extraction
	Forest intensification	Stable forest land cover, increase in wood extraction

We compared priority areas under current and future land use, separately for the A1 and B2 scenario, and classified them into three groups as only priority in 2000 ("current only"), only priority in 2040 ("future only") or priority in both 2000 and 2040 (stable). We combined the three groups of priority areas with the land use change process in each location to analyze the relation between land use changes and changes in top priority ranking. Following Stürck et al. (2015a), we identified land change trajectories based on the change in land use between 2000 and 2040, but made some minor adjustments for the purposes of this paper (Table 5.2). Land change trajectories are hierarchically based on changes in land cover followed by changes in management intensity (Stürck et al., 2015a). Land management intensity for croplands (fertilization), grasslands (stocking density) and forests (wood removal) was quantified using a continuous scale. The continuous management intensity values were reclassified into three management intensity classes for cropland (low, medium, high) and two management intensity classes for grasslands (low, high) based on previously determined threshold values (Stürck and

Verburg, 2017). For cropland and grassland sites, intensification or extensification (i.e. a reduction of management intensity) was recorded when the intensity class changed over time. For forests we used the original continuous management intensity classification and changes in land management intensity (intensification vs. extensification) were recorded based on a 5% change in wood removal between 2000 and 2040. As described in section 2.2, the changes of ESs over time depend on changes in land cover and land management intensity.

For the assessment of ESs change in priority areas, we analyzed two different metrics of change in ESs. First, we calculated the % of each ES maintained for increasing percentages of land assigned as priority area. A comparison across the full priority ranking provides information on how ESs in priority areas are affected by land use change and to what extent changes in ESs are reflected across the full ranking. Second, we calculated the mean change in ES capacity for priority areas at the regional level, using NUTS3-regions (Eurostat/GISCO, 2011). NUTS3-regions are statistical units for the whole EU largely following administrative boundaries in which population ranges from 150,000 – 800,000.

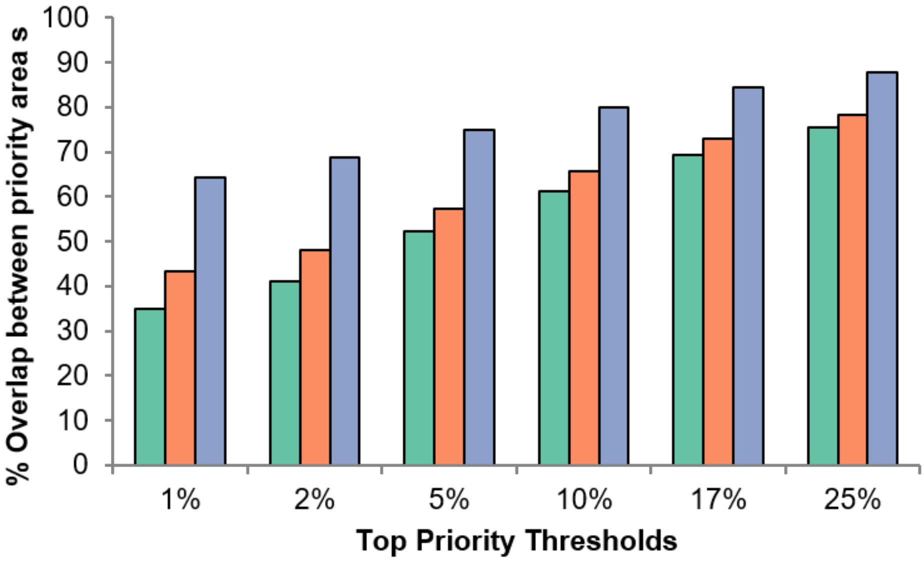
5.3 Results

5.3.1 Shifting priorities for ecosystem services

Both land use change scenarios result in shifts in top priority areas over time (Figure 5.2). Here we primarily focus on the top priority areas but see the S19 for a full priority ranking per scenario. The congruence between current priority areas and future A1 priority areas ranges from 34.8% for the top 1% priority areas to 75.4% for the top 25% priority areas. Thus, for a priority network of around 40,100 km^2 (top 1%) only 13,900 km^2 retains its priority rank under both current and future land use, while 26,200 km^{2-} is no longer a top 1% priority area due to land use change. For the B2 scenario the contrast is not as strong (43.3% overlap with current in top1%), but this still means the majority of sites is no longer a top 1% priority site due to land use change (Figure 5.2 orange bars). Priority area sets identified under projected land use for 2040 (A1 vs. B2, Figure 5.2 blue bars) have much more overlap indicating that differences in land use change among scenarios have a smaller effect on shifting priorities.

Priority areas shared between all three prioritization experiments are spatially clustered and are predominantly located in the Mediterranean and in mountainous regions such as the Alps, the Carpathians and the Pyrenees. Priority areas unique to a specific experiment are much more dispersed and scattered across Europe without clear spatial patterns (Figure 5.3). We identified three sets of shifting priorities: i) priority areas at risk, i.e. those priority areas unique to the year 2000 land use prioritization; ii) new priority areas, i.e. those priority areas unique to the year 2040 land use prioritization (A1 or B2); and iii) stable priorities, i.e. priority areas shared between two or three prioritizations of the years 2000 and 2040. In general we observe two generic patterns of priority area distribution. First, we see areas where both stable and non-stable priority areas are clustered. Here the non-stable priority areas tend to be located at the edges of larger patches of stable priority areas (panel C in Figure 5.3). This pattern is common around all locations with larger stable priority areas. Second, we see areas where both stable and non-stable priority areas are completely scattered without any spatial patterns (panel A in Figure 5.3). This pattern is common in the absence of larger patches of stable priority areas and occurs predominantly in northern European countries.

Figure 5.2: Degree of overlap in priority areas for ecosystem services between the prioritization experiments. Results are presented for varying percentages of land assigned as priority (x-axis). The percentage overlap between priority areas is for the experiments 2000 and 2040 (A1) (green), 2000 and 2040 (B2) (orange) and 2040 (A1) and 2040 (B2) (blue).



Changes in priority areas at the cell level are only partly reflected in regional changes. Out of the 1296 NUTS3-regions 1071 primarily have stable priority areas, even in regions where the stable priority areas are scattered (Figure 5.4). However, 60 NUTS-regions have a relative high share of priority areas that are identified as such only in the year 2000. These regions are located in Northern Scotland, Central France, Southern Fenno-Scandia and the Baltic states and especially in Poland and Czech republic. Here land use change has predominantly negative effects on ESs, resulting in a concentration of areas that initially are prioritized but are likely to forego priority under land use change in the absence of conservation measures. Only 13 NUTS-regions had a high share of new priority areas following land use change. The lower number of regions with new priority areas compared to the number of regions with foregone priorities indicates that newly established priority areas are not concentrated in specific regions.

The top priority areas are characterized by only a limited set of land use change processes (Figure 5.5). We cannot identify one dominant land use change process across all three sets. Forest management intensification strongly overlaps with the location of priority areas at risk (33.5 % or 69,620 km^2 of priority areas unique to the year 2000 land use). Forest management intensification negatively effects both carbon sequestration and flood control. Other ESs are not affected by forest management intensification, which possibly explains why a fraction of top priority areas remains stable, despite forest management intensification (17.4% or 86,312 km^2 of stable priority areas). New priority areas are largely characterized by land abandonment (17.0% or 35,329 km^2 of priority areas unique to the year 2040 land use) and forest succession (32.0% or 66,503 km^2 of priority areas unique to the year 2040 land use). Dominant land use change processes affecting priority areas are thus forest intensification, land abandonment and forest succession whereas other land use processes do not show clear patterns with the shifts in priority areas.

5.3.2 Changing ecosystem services over time and space

Land use change shifts priority areas over time but it only results in declines of ES capacity for a limited set of ESs (Figure 5.6). For the entire EU land use change results in declines of carbon sequestration and pollination whereas erosion control, flood regulation and nature-based tourism are stable or show minor increases. This general pattern is also reflected in the top priority areas: future top priority areas have lower pollination and carbon sequestration capacity than current top priority areas. Pollination declines stronger at the EU scale(-18.0%) than at the scale of the top 17% priority areas (-14.3%). In contrast, carbon sequestration declines stronger at the scale of the top 17% priority areas (-17.4%) than at the scale of the EU (-16.5%). This means that the shift in priority areas could partly compensate for the losses in pollination capacity, but was unable to compensate for the losses of carbon sequestration capacity in Europe. Land use change thus results in shifts in priority areas but the future capacity is only decreasing for a limited set of ESs.

Figure 5.3: Congruence in top priority areas (17%) between the three prioritization experiments. The left depicts the overlap in top 17% priority areas between the three experiments for Europe. The two right panels show the congruence of top 17% priority areas for two regional cutouts. Stable priorities are priority areas that are at least shared between two of the three experiments.

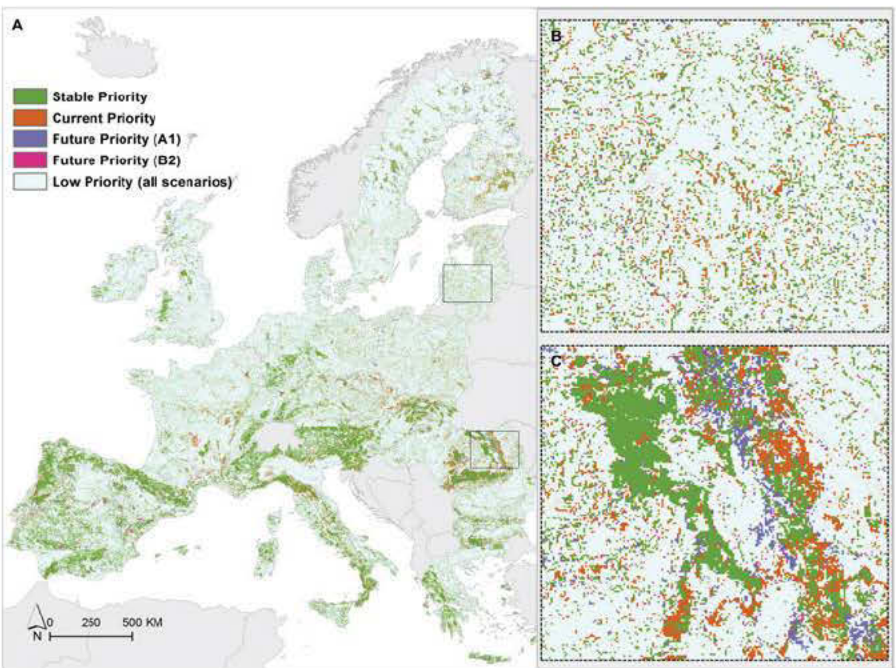
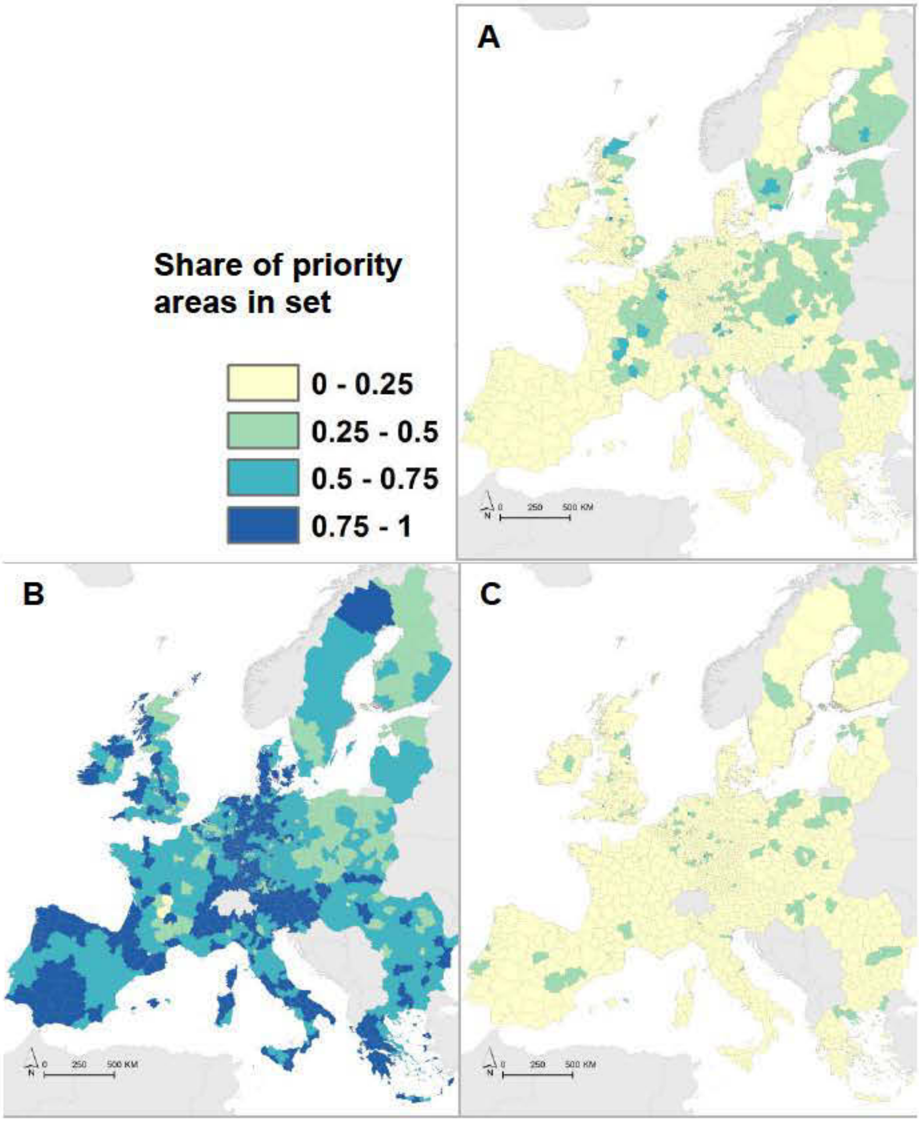


Figure 5.4: Share of priority areas per administrative unit (NUTS3-region) assigned to a specific class of priority areas namely: (A) priority areas only under current land use, (B) stable priorities under both current and future land use (A1 experiment) and (C) priority areas only under future land use (A1 experiment).



Although the total capacity of three out of five ESs remained stable across priority areas in Europe (Figure 5.6), at the regional level, land use change affected ES capacity (Figure 5.7). In almost none of the NUTS3-regions ES capacity was stable over time within their priority areas. Most commonly, land use change results in trade-offs between ESs at the regional level and these trade-offs occured all across Europe. In other regions land use change results only in increases or decreases in ES capacity. These regions tend to be spatially clustered: decreases in ES at the regional level are found in Eastern Europe and in France. Eastern Europe, especially Poland and Czech Republic, are characterized by large areas with forest and cropland intensification. ES capacity tends to increase in many Spanish regions, southern England and in Denmark characterized by forest expansion and decreasing management intensity of cropland and pastures. Land use change is thus driving changes in ES capacity at the regional level all across Europe and regional changes in ES capacity do not necessarily translate into changes in ES capacity at the EU level.

Figure 5.5: Overlap between land use change and three sets of priority areas, namely (A) priority areas only under current land use, (B) stable priorities under both current and future land use (A1 experiment) and (C) priorities only under future land use (A1 experiment). The area of the respective pie charts depicts the difference in land area. Results are presented for the top 17% priority areas.

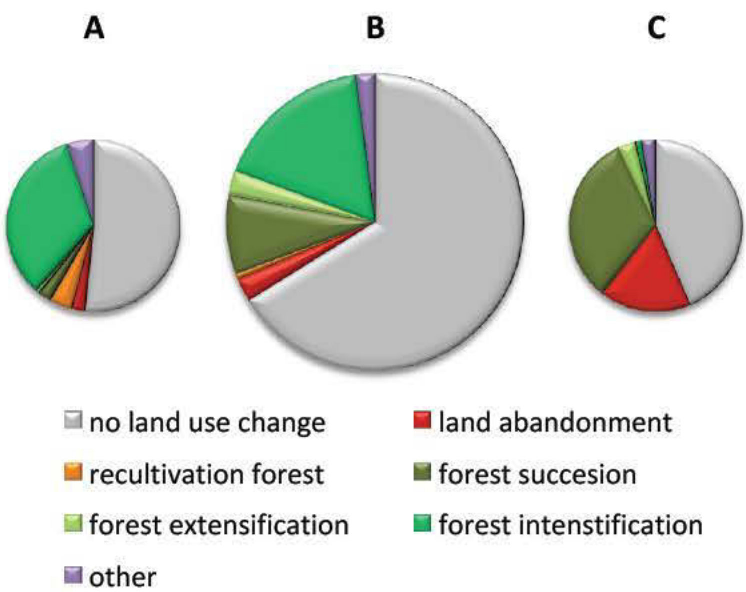
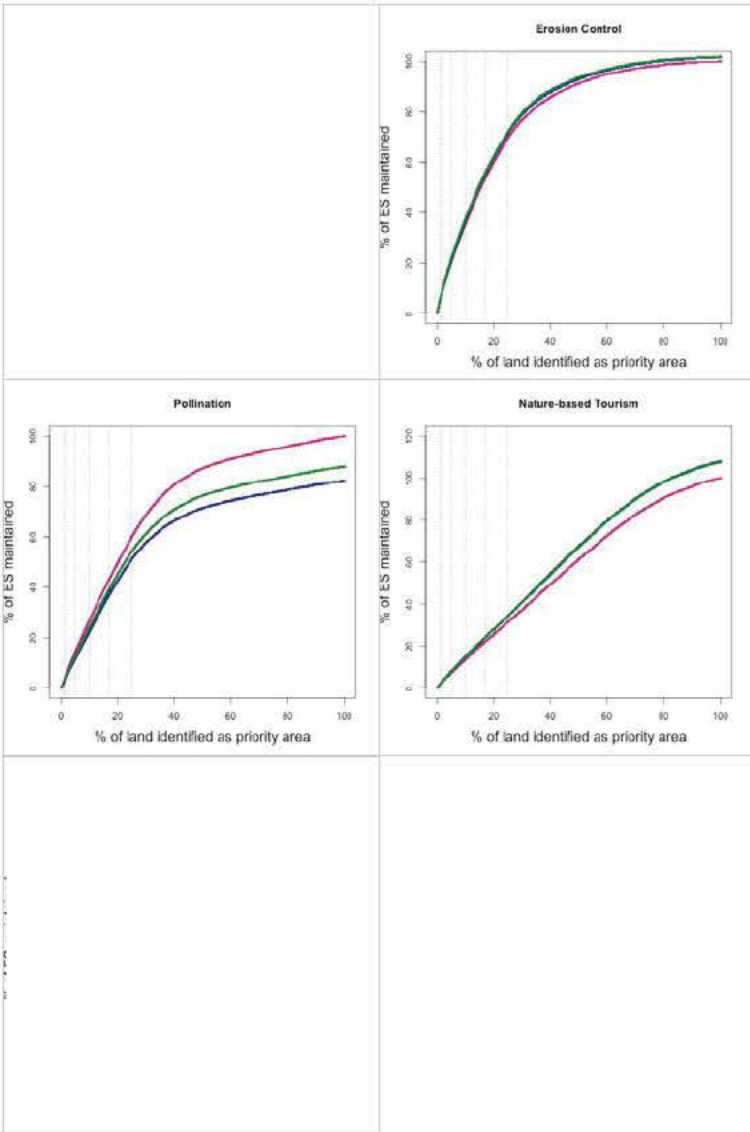


Figure 5.6: Percentage of ecosystem service (ES) maintained with increasing % of land assigned as priority area. The level of ES maintained is relative to the total amount of ES under current land use (2000), meaning that the lines for future land use (scenarios 2040 (A1) and 2040 (B2)) can exceed the 100% ES maintained (as is the case for nature-based tourism). The more concave the graph the smaller the fraction of land required to maintain a high % of that particular ES. Vertical dashed lines indicate the top priority levels ranging from 1% to 25%. Although the prioritization is performed for all ES combined, results are presented for each ES separately to highlight the differences between the ESs.



5.4 Discussion

5.4.1 Reflection on the results

In this study we prioritized areas in Europe for ESs. We did so for current and future land use based on results of dynamic models for land cover change, land management change and ESs over time. An important finding is that land use change resulted in shifts in the location of priority areas over time. These shifts in the location of priority areas can have different causes. For a number of services, i.e., pollination and nature-based tourism, not just local land use change, but also land use change in the neighborhood can affect local ES capacity. Especially for pollination cropland abandonment and cropland expansion can strongly affect where ESs are supplied locally because of the required proximity between natural habitat and cropland areas for pollination. In a German agricultural landscape changes in ES capacity could not be fully explained without incorporating changes in landscape configuration besides changes in local land cover (Lautenbach et al., 2011). Effects of landscape configuration could be partly accounted for by connectivity measures in Zonation (Kukkala and Moilanen, 2017). However, for many ESs there is still limited knowledge on how configuration affects ES capacity (Verhagen et al., 2016). For all services local land use change changes the local ES capacity but because all results are nested it also changes the relative priority of other areas. For the prioritization analysis non-local changes in ES capacity can affect both the relative ranking of sites and the complementarity value of sites regarding ESs even without any change in ES capacity over time at the site itself. Accounting for flow zones partly accounts for this issue because only changes in ES capacity over time only affect the relative value and complementarity of sites within a flow zone for that particular service. To conclude, non-local changes in land use may affect the prioritization of an area 1) by changing its ES capacity through configuration effects, or 2) without changing its actual ES capacity, but because the ES capacity of an area is always relative to the ES capacity of other areas.

Besides land use change also forest aging affected the priority area allocation. For carbon sequestration forest aging affected the sequestration capacity over time. This effect is relatively limited because forest aging could only explain a 4% decrease in carbon sequestration over time without land use change (Schulp et al., 2008).

These interacting processes make it difficult to determine the causes of changes in priority at specific locations. In some areas changes in priority occurred while land use was stable while in other areas land use change did not affect the priority assigned. Nevertheless, from the overlay analysis it is clear that forest intensification is an important reason why areas of high priority under current land use are no longer top-ranked in 2040. In contrast, expansion of high intensity cropland (+12% in area in B2 scenario) does not directly affect current priority areas for ESs because these were already mostly located outside croplands. At the same time, agricultural abandonment and forest succession are important factors explaining new priority areas.

Previous studies have mostly looked at the impact of land cover change on ESs (Metzger et al., 2006; Stürck et al., 2015b; Polce et al., 2016). In the study of Polce et al. (2016), a set of three regulating services showed only minor changes in ES capacity between 2000 and 2050 for most areas in Europe. In two different studies, carbon sequestration capacity over time is expected to slightly increase as result of land cover change, especially agricultural abandonment (Schulp et al., 2008; Stürck et al., 2015b). The limited or negative impact of land use change on the aggregated ES capacity in our analysis can be explained by the fact that we used a different set of ES models and because we fully accounted for changes in land management in addition to changes in land cover. Previous research already highlighted that different methods to map ESs across Europe also result in different ES maps (Schulp et al., 2014a). This applies to current assessments of ESs but is also likely to hold for future predictions of ESs. More importantly, irrespective of the changes in aggregated ES provisioning at EU level, all studies find strong local and regional differences in the effect of land use change on ES capacity (Alcamo et al., 2005; Schulp et al., 2008; Stürck et al., 2015a; Polce et al., 2016). In other words, although the exact changes in ES capacity differ somewhat between studies, the main finding of this study - that land use change will result in shifting priorities driven by local changes in ESs - is likely independent of the trends in ES capacity over time and the choice for particular ES models.

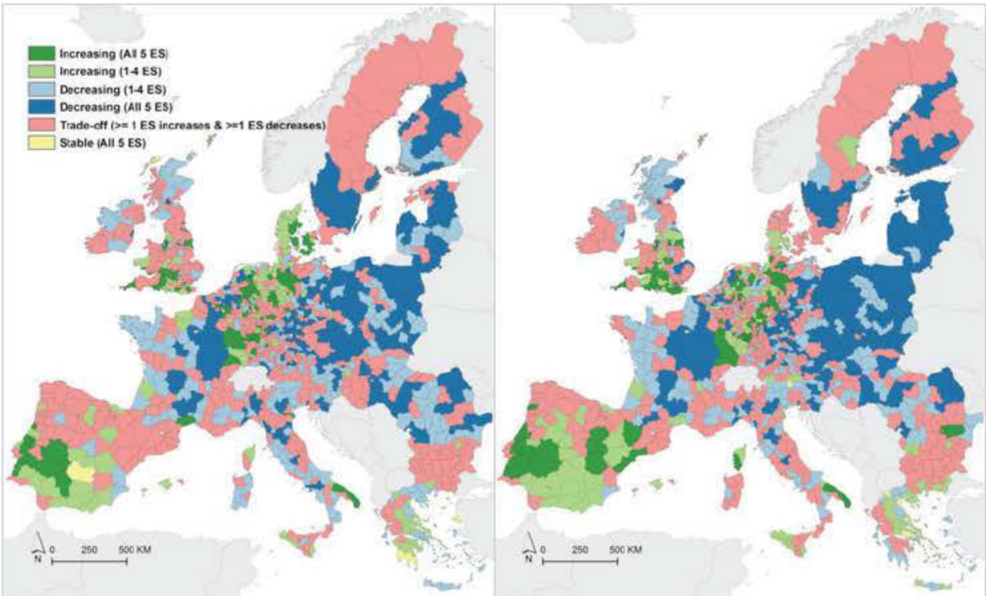
Within priority areas land use change resulted in both positive and negative changes in ES capacity over time at the local (cell), regional and EU level. Earlier studies seldom accounted for land use change directly, but rather assessed it as a threat to identified priority areas using negative weights (Luck et al., 2012; Cimon-Morin et al., 2016). In a study for a remote region in Canada expansion of urban areas resulted in decreasing ES capacity within priority areas over time (Cimon-Morin et al., 2016). In such circumstances, postponing the identification of a protected area network would lower the ESs over time (Cimon-Morin et al., 2016). When multiple land use change processes are included the effects on ESs are more diverse, with local increases and decreases. For example, in a study on a watershed in Japan, Fan et al. (2016) found no to slightly positive effects from land use change on ESs, with hardly any impact on the associated locations of priority areas. From this and our study we can conclude that accounting for the full array of possible land use change processes locally results in both positive and negative effects of land use on ESs without necessarily threatening ES capacity for the entire study region. Hence, in contrast to the common approach in prioritization studies it is not possible to assign uniform negative effects of land use change on priority areas for ESs because such an approach does not account for differing impacts of land use change on ES depending on the land use change, the ES studied and local positive and negative effects of land use change based on spatial configuration.

5.4.2 Reflection on the methodology

We used a straightforward approach to account for the effects of land use change on ES prioritization. Because of our primary interest in the effects of land use change we did not account for a number of other factors important for prioritization for planning purposes, such as non-uniform land costs. Consideration of land costs can also increase the efficiency of conservation networks (Kark et al., 2009). Here we used area as a uniform cost measure which is a common proxy in ES prioritization research (Remme and Schröter, 2016; Verhagen et al., 2017). Land costs can be approximated using different measures including land prices, management costs and opportunity costs of foregone production (Naidoo et al., 2006). Threats including land use change are sometimes used as a surrogate for costs but should preferably be treated separately (Naidoo et al., 2006). Land costs are likely to change over time, partly as a result of changes in land management, land cover and societal demand. However, how these costs will change in the future is unknown. Therefore, we decided to use a uniform cost measure over space and time and only assess the direct effect of land use change on ES capacity.

We prioritized land in Europe based on the occurrence of five ESs. Although being a limited set, the ESs selected do cover the full array of possible responses to land use change with some being only sensitive to changes in local land cover, some being sensitive to changes in local and surrounding land cover and some being sensitive to land management. Conservation management for biodiversity and ESs especially in production landscapes should incorporate all these facets of land use change (Seppelt et al., 2016; Verhagen et al., 2016).

Figure 5.7: Projected change in all ecosystem services within priority areas per administrative (NUTS3)-region. Panel A (B) shows the change in ecosystem services under current and future land use for experiment A1 (experiment B2). An ES is considered to increase (decrease) over time if the difference between current and future ES supply is at least 5%. Results are presented for the ES capacity within the top17% priority areas



In our analysis we only focused on the top ranked priority areas as is most common in prioritization studies. In recent years prioritization studies have also looked at the full ranking of a landscape or used prioritization to identify areas most suited for development based on the lowest ranked areas (Kareksela et al., 2013; Nin et al., 2016). The focus on top priority areas is especially important from a conservation perspective in relation to threats from land use change. We found that current priority areas were mostly affected by forest management intensification and new priority areas had undergone land abandonment or forest succession. For the whole of Europe mean priority rankings were more strongly driven by changes in land cover compared to land management (S19). For example, the conversion of pastures and forests to cropland had a strong negative effect on mean priority rankings but hardly impacted top priority areas. In contrast, forest management intensification across the full priority ranking had almost no effect on the mean priority rank. This finding shows that it is important to distinguish between land use change processes affecting the full priority ranking versus land use change processes affecting priority areas.

In our analysis we did not account for impacts of climate change on ESs over time. Recent studies looked at the combined effect of land use and climate change on ES capacity (Polce et al., 2016) and on ES prioritization (Fan et al., 2016). In Europe land use change had a stronger effect on ES capacity over time compared to climate change (Polce et al., 2016). Moreover, the timeframe of our study is relatively short. On longer time scales the relative impacts of climate change as compared to land use change may be larger.

Land use and land use change models at the European extent are never validated due to inconsistencies between datasets and large uncertainties in the most important land change processes in the Corine Land Cover multi-year data. However, the model application used in this study was part of a inter-model comparison of land change models at European extent (Alexander et al., 2017). This effort showed large variations between European land use models, with the DynaCLUE model falling within the range of that variation (Alexander et al., 2017). For smaller scale applications the DynaCLUE model has often been validated and generally has a high validation score (e.g., Pontius et al 2008). In spite of this, the work presented here should be interpreted with care and be interpreted in terms of the variation in the effect of land use change on priority areas for ESs under two contrasting scenarios of land abandonment.

Our study has, indirectly, accounted for the demand of ESs through the use of flow zones. Previous studies have highlighted that changes in ES demand are likely larger than changes in ES capacity over time and might result in increased mismatches between areas of high ES capacity and areas of high ES demand (Alcamo et al., 2005; Stürck et al., 2015a). Whether changes in priority areas over time are more strongly driven by changes in ES capacity or by changes in ES demand remains an important question for future research with possibly important consequences for the management of ESs.

5.4.3 Implications for management

There is a need to operationalize the concept of ES in landscape planning and decision making (de Groot et al., 2010; Balmford et al., 2011). In general, the identification of priority areas cannot be easily translated into management practices. Identification of priority areas is often part of a wider framework of systematic conservation planning (Moilanen et al., 2009). The aim is to identify the most cost-effective protected area network where management for ES would focus on protection alongside biodiversity conservation. Whether the maintenance of ESs requires the designation of protected areas, or other measures, depends on the nature of the threats to the services.

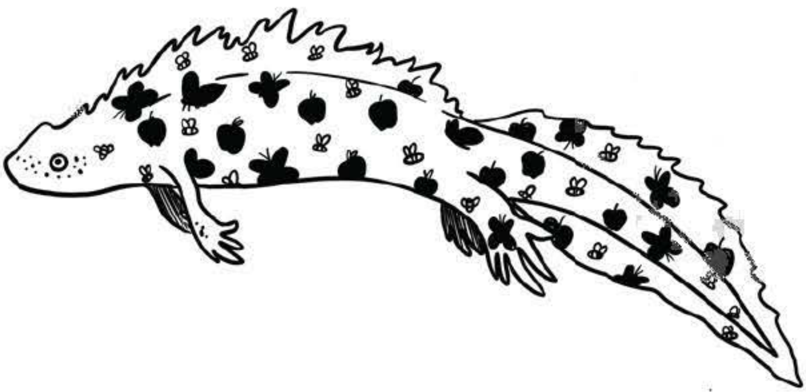
However, for many ESs a certain use of the area is required to obtain benefits from ESs. According to some authors, the ES selected for prioritization should be compatible with biodiversity conservation objectives (Chan et al., 2006; Schröter et al., 2014b). These areas should then be managed for a sustainable use of the area where benefits obtained from ES are not threatening biodiversity conservation. However, knowledge on what levels of use are compatible with biodiversity is missing and limiting use of ecosystems might be in direct conflict with meeting certain demands for these services requiring an expansion of sites providing ESs. Land management should aim to navigate these potential trade-offs regarding the conservation of biodiversity and ESs (Cimon-Morin et al., 2013).

A better understanding of the scale and impacts of land use change can provide guidance for the management of priority areas. Stable priority areas tend to be clustered in remote mountainous regions of Europe. Here the identification of priority areas may not need to result in the designation of protected areas. The real management challenges lies in minimizing land use impacts on ES capacity in more intensively-used landscapes i.e. where production and other usage of the land co-occur and are often competing (Schulp et al., 2016). Here ESs are often supplied by relatively small patches of semi-natural vegetation. Similar for biodiversity, in these areas pressures of land use change, including intensification, are highest (Seppelt et al., 2016). Here the protection of patches providing local services might be required. In Europe, the Natura 2000 network is the most important land protection scheme for biodiversity with a relatively good performance for biodiversity protection (van Teeffelen et al., 2015a; Kukkala et al., 2016). A simple overlay of the current Natura 2000 network (EEA, 2016) with priority areas that are at risk of land use change, indicates that around 75% of the priority areas at risk are found outside this network (not shown). This suggest that the current Natura2000 network may need extension if it is to protect most of the current priority areas for ESs that are at risk of land use change. To test the effectiveness of protected area measures for ESs future research could develop scenarios where priority areas for ESs are coupled to land use simulation models, i.e. allowing no land use change in the top current priority areas for ESs.

Land use change processes, especially agricultural land abandonment and forest succession, resulted in the establishment of new priority areas. Land use planning and management could prioritize these areas to develop into a way that maximizes the gains in ES capacity. This coincides with the notion that conservation should not only aim at the protection of current priority areas but also stimulate favorable development/restoration of new areas with high potential. In Europe this relates especially to agricultural abandonment areas that can only achieve their potential upon appropriate management of re-wilding (Ceausu et al., 2015; Schulp et al., 2016).

Our results indicate that avoiding land use change at a single location is not sufficient to maintain the same level of ESs due to neighborhood interactions, requiring a landscape-scale approach (Mitchell et al 2015; Verhagen et al 2016b). Moreover, many ES need to be supplied at a local or regional scale requires that priority areas are distributed across flow zones (Verhagen et al., 2017). While techniques are being developed to account for connectivity and minimum area requirements for ESs (Kukkala and Moilanen, 2017), these two factors suggests the importance of an integral land management approach.

Our study also found regional changes in ES capacity within priority areas over time (Figure 5.7).Assessments and management of ES over time should therefore not only look at the EU wide impacts of land use change on ES capacity but also account for changes at the regional level. Many ESs have a regional or local flow with benefits being obtained locally or regionally, which makes this a logical management scale for this type of ESs. For most ESs, local losses in ESs capacity cannot be offset by distant gains. For biodiversity, non-coordinated management at smaller spatial scales to prioritize areas is known to result in sub-optimal outcomes (Kark et al., 2009; Pouzols et al., 2014). Further developing techniques to balance between the most economic outcome and the need to maintain most ESs at smaller spatial scales is a crucial step to effectively identify priority areas in a context of land use change.



6. Agri-environment measures to restore biodiversity and ecosystem services in an agricultural landscape

Abstract

Demands on peri-urban landscapes are increasing and diversifying. These landscapes typically fulfil different functions, including agriculture, ecosystem services and may also host species and habitats of conservation concern. Designing landscapes that can simultaneously meet multiple competing demands is an important challenge. Addressing this challenge requires methods that can provide a clear understanding of the trade-offs between biodiversity, production and ecosystem services, and that can assist in effectively navigating these through planning. Here, we tested the degree to which landscape optimization algorithms can do so, for an intensively-used area in the Netherlands. We optimized land use/land management to increase fruit yield, endangered species habitat, and landscape aesthetics, while minimizing losses in dairy farming, and assessed the trade-offs among these objectives. We considered the allocation of on-farm measures (organic management and establishment of linear elements), off-farm measures (taking land out of production) and a combination of both. Both agri-environment measures were able to contribute to the objectives but showed strong trade-offs between fruit yield (on-farm: +26.19% vs. off-farm: +1.63%) and species habitat (on-farm: +9.90% vs. off-farm: +45.72%). Using a combination of both on-farm and off-farm measures largely alleviated this trade-off. The spatial allocation of measures in the landscape was important, and priority areas according to our optimization technique differed markedly from those in the existing nature conservation plan, which is primarily focused on species conservation. Our results highlight that the current nature conservation plan can be improved, thereby simultaneously contributing to multiple environmental objectives while incurring a smaller impact on dairy farming. Comparing *on-farm* and *off-farm* management practices provides insight in the functional trade-offs associated with each management option and their respective potential to increase multifunctionality. Moreover, the identification of priority locations across all solutions can further integrate landscape optimization approaches into spatial planning and inform policy design and implementation.

This chapter is published as:
Verhagen, W., van der Zanden, E.H., Strauch, M., van Teeffelen, A.J.A. and Verburg, P.H. (2018), Optimizing the allocation of agri-environment measures to navigate the trade-offs between ecosystem services, biodiversity and agricultural production. Environmental Science and Policy 84: 186-196. doi: 10.1016/j.envsci.2018.03.013

6.1 Introduction

Human demands on landscapes are multifold and these demands often compete for the same space. Agricultural landscapes have often been optimized for the production of food, resulting in declines of both biodiversity and non-provisioning ecosystem services (Bennett et al., 2009; Seppelt et al., 2016). However, with increasing human population size and peri-urban development the multitude of demands on these landscapes often increases (Zasada et al., 2013). To meet multiple demands in the future, many studies suggest that agricultural landscapes should become multifunctional (e.g. O'Farrell & Anderson 2010; Fischer, Meacham, et al. 2017). A shift towards a more multifunctional landscape may require changes in farm management and nature restoration (Tscharntke et al., 2005, 2012). However, such shifts inevitably involve trade-offs between conflicting objectives (Howe et al., 2014; Fischer et al., 2017a, 2017b). Understanding and balancing these trade-offs therefore has a high priority on the policy agenda. Current trade-off research needs to move beyond the identification of trade-offs towards the development of tools that can assist landscape planners in effectively navigating these trade-offs, e.g. by supporting target setting based on alternative 'optimal' management strategies (Seppelt et al., 2013; Bennett et al., 2015; Verburg et al., 2016).

In the presence of such trade-offs, optimization algorithms are capable of identifying a set of Pareto-optimal land use and land management (LULM) configurations (Nelson et al., 2009; Lautenbach et al., 2013; Gourevitch et al., 2016; Kennedy et al., 2016; Pennington et al., 2017). Previous analyses have shown that trade-offs not only exist between agricultural production and regulating ecosystem services, but also between individual ecosystem services themselves (Nelson et al., 2009; Howe et al., 2014; Gourevitch et al., 2016; Kennedy et al., 2016). Optimization algorithms can provide insight into the functional trade-offs between two or more objectives and provide a full set of possible future LULM allocations (Lautenbach et al., 2013; Seppelt et al., 2013; Cord et al., 2017). Optimization algorithms can therefore depict the effects of landscape management options for multiple objectives simultaneously, and provide alternative pathways for balancing these trade-offs (Seppelt et al., 2013; Verburg et al., 2016; Cord et al., 2017). Furthermore, optimization approaches hold great potential for bridging the science-policy divide by comparing current conservation plans and alternative scenarios to the full set of alternative future LULM allocations (Seppelt et al., 2013; Cord et al., 2017).

A landscape's multifunctionality can be increased using a diverse set of LULM options such as restoration of natural areas or changes in farm practices (Batáry et al. 2015; Seppelt et al. 2016; Duru et al. 2015; Lovell & Johnston 2009). In addition, green linear elements, such as hedges and tree lines, are capable of providing multiple ecosystem services and hold great potential for landscape optimization in agricultural areas (Jones et al., 2013; Kremen and M'Gonigle, 2015; Verhagen et al., 2016). Policy instruments to increase the multifunctionality in European landscapes also cover this full range, from policies mostly focused on changes in farming practices through the EU Common Agricultural Policy (rural development and agri-environment measures) to policies focussed on restoration of green infrastructure.

Previous landscape optimization analyses have either focused on restoration of natural areas (*off-farm*) (Nelson et al., 2009; Gourevitch et al., 2016; Kennedy et al., 2016) or on allocating a set of farm management alternatives (*on-farm*) (Lautenbach et al., 2013; Pennington et al., 2017). Previous research further showed the potential of optimization algorithms in minimizing trade-offs between forestry, biodiversity and ecosystem services, following forest restoration in Uganda (Gourevitch et al., 2016) or optimizing crop rotations schemes for food production, biodiesel crops and river management (Lautenbach et al., 2013). However, *on-farm* and *off-farm* management practices have hardly been compared nor combined in landscape optimization analyses limiting our knowledge on the functional trade-offs associated with each management option and their respective potential to increase multifunctionality.

6

This paper presents a multi-objective landscape optimization for *on-farm* and *off-farm* agri-environment measures for the Kromme Rijn area, The Netherlands. The Kromme Rijn area is an agricultural landscape dominated by pasture production, rich in green linear elements. We compare landscape optimization for *on-farm* and *off-farm* agro-environment alternatives for indicators of production, biodiversity and ecosystem services. We compare our outcomes to the current nature conservation plan to assess possible improvements of that plan with respect to the values per objective and the priority locations for agri-environment measures.

6.2 Methods

The method section consists of four parts. We first provide the background of the study area and the current nature management. We then describe the spatial data used in this study. Third, we present the models used to quantify the environmental objectives and fourth, we describe the optimization method.

6.2.1 Case study background

The Kromme Rijn area (Figure 6.1) is a peri-urban agricultural dominated landscape of 219 km^2, located in the Province of Utrecht, The Netherlands (52°06'28.1"N - 51°57'17.6"N; 5°06'10.3"E - 5°37'10.2"E) (OKRA and Provincie Utrecht, 2011). The landscape is characterized by an old tributary of the river Rhine, the Kromme Rijn, meandering through the area. The agricultural landscape in the area is rich in woody linear elements and characterized by a NW-SE gradient in openness. In the northern section, the banks of the Kromme Rijn are flanked by estates dating back to the 19th century. Pasture production around these estates is intertwined with remnants of natural vegetation and bordered in the north by forests. In the rest of the area the main agricultural activity is dairy farming combined with fruit yield on sandy and clay levee deposits of the former riverbed. In addition to its agricultural function, the area has high aesthetic quality and is an important recreational area for the city of Utrecht (Geerdes et al., 2016; Utrecht Province, 2017).

A nature conservation plan (NCP) was developed by the province to enhance species conservation in the Kromme Rijn area. The NCP focuses on the restoration of habitat for a set of focal species, including the great crested newt, the common noctule or the long eared owl (Utrecht Province, 2017). In the NCP, agricultural land is assigned to be taken out of production and typically restored towards natural grassland. In addition, the NCP promotes the establishment of green linear elements, such as hedges and tree lines, on agricultural land. Establishment of these elements is a voluntary measure but is eligible for financial support. A set of stakeholder workshops (March 2016 and December 2016) revealed that farmers consider the transition to organic farming as an important alternative to conventional farming, given recent high prices for organic milk. The transition to organic management is not considered in the NCP and is not eligible for financial support. The NCP does not consider additional environmental objectives important for the Kromme Rijn area, such as its aesthetic landscape value or fruit yield. Especially of interest are the traditional/extensive orchards, which support biodiversity and ecosystem services besides providing fruits.

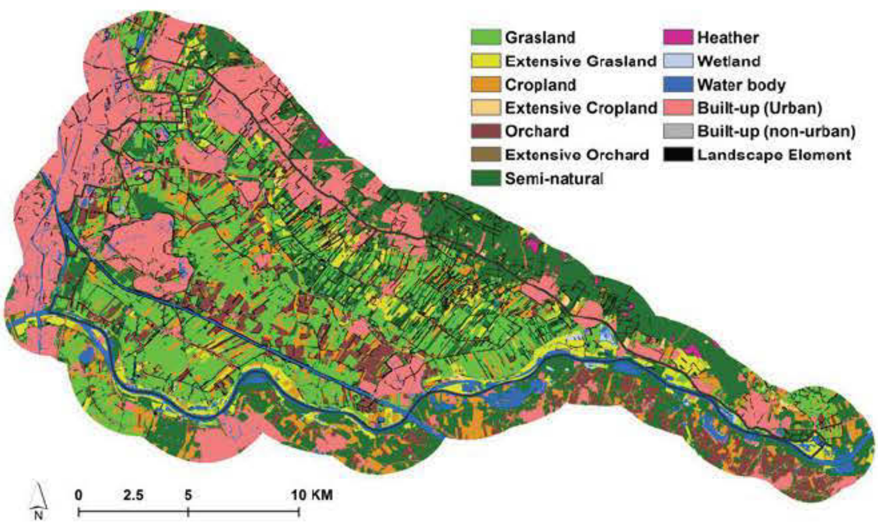
6.2.1 Spatial information - Land use and land management map

We made a land use land management (LULM) map at 25x25 meter resolution based on a combination of land cover, crop types, management intensity and the location of linear elements, yielding 41 different land use types (S21). All agricultural areas were assigned a combination of land management and presence of linear elements, resulting in four categories: conventional without linear elements, conventional with linear elements, organic without linear elements and organic with linear elements.

For the delineation of organic and conventional management for orchard and pasture fields, we developed a map of organic agriculture based on the addresses of all organic farms, which we obtained from the control agency of organic farming (SKAL, 2017). We assigned organic management to crop fields based on the location of organic farms, their main crop production type, average farm size per production type and a spatial data set of crop types per field (S22). Fields closest to an organic farm were assigned organic management until the area of the fields equalled the average farm size. Within a field, all cells were either under conventional or organic management. Information on the management of natural areas was obtained from the most recent nature conservation plan (NCP; Utrecht Province 2017).

In addition, we identified the presence of linear elements in cells, using a detailed dataset on the location of tree lines and hedges (Utrecht Province, 2013). The extent of linear elements is commonly smaller than the resolution used to map dominant LULM types per cell. Therefore, we determined for each cell whether linear elements were present and subsequently categorised each LULM into two categories; with or without these elements. A detailed methodology behind making the LULM map and a list of all LULM types is provided in the S21.

Figure 6.1: Land use land management map of the study area depicting the main agricultural land systems. The inset of the map shows the location of the Kromme Rijn area (in red) within The Netherlands. The land system map is depicted with a 2 km buffer. The number of classes (41) in the actual map is simplified for visualization purposes.



6.2.2 Environmental objectives

We modelled three environmental objectives, namely fruit yield, aesthetic landscape value and habitat suitability for great crested newt (*Triturus cristatus*). As the implementation of agri-environment measures comes at a cost of pasture production, we quantified the loss in pasture production, whose minimization served as a fourth objective. The four objectives were selected based on relevance to the study area using input from stakeholder workshop, the nature conservation plan and other reports (Geerdes et al., 2016; Utrecht Province, 2017). Also, they represented different types of objectives for landscape planners, namely agricultural production, a cultural ecosystem service and a biodiversity indicator. The models used to quantify each objective are described in full in the S22. Below we describe the main characteristics. All the environmental models were written in R (R Core Team, 2016).

Pasture production for dairy cows

The model of pasture production for dairy cows (euro/ha/year) was based on a look-up table approach. We calculated the profit per cell based on average production values per ha pasture, the costs of milk production and market milk prices for the Netherlands (agrimatie.nl, 2014). All data were averaged over the years 2010-2014. We used animal feed as the cost measure, since it is an important cost in switching from conventional to organic farming (SKAL, 2017).

We accounted for pasture management by using different production quantities, market prices and costs for organic and conventional pastures. We also accounted for a transition period of six months for farmers switching to organic management. During this transition period farmers already implemented organic management on the field, which leads to lower production outputs and higher costs, but the products are still sold for conventional market prices (SKAL, 2017).

The presence of linear elements lowered the amount of land in production per cell. In all models we used hedges to approximate the effects of linear elements, with a standard width of 5 meters. This width is within estimates of previous studies using hedgerow widths varying between 3 to 10 meter (Van Teeffelen et al. 2015b; Schulp et al. 2014).

We calculated the total profits of pasture production per cell for a period of ten years to be able to include transition costs to organic farming. Given the current price-cost ratio, conventional management is most profitable and therefore both a transition to organic farming or restoring linear elements incurred a financial loss. We therefore assumed a reduction in pasture production as an opportunity cost of foregone production. To estimate this opportunity cost, we first calculated the maximum pasture production, i.e. the pasture production if all pastures were assigned to conventional management without linear elements. Costs of foregone production were then calculated as the pasture production with the newly assigned LULM allocation minus the potential pasture production.

Fruit yield

Fruit yield (euro/ha/year) was modelled based on the level of pollination per orchard coupled with a look-up table approach to quantify the costs and benefits. Fruit tree production partly depends on pollination for fruit set. Pollination by wild bees and other pollinators is more effective and more resilient against diseases than pollination by domesticated honeybees (Potts et al., 2010; Garibaldi et al., 2013). We therefore adopted an existing pollination model by Zulian et al. (2013), linking the landscape suitability for wild bees, based on nest suitability, floral resources and distance to orchards. In a second step, we calculated the fruit yield based on estimates of production, costs and market prices for apples and pears, which are the most dominant fruit crops in the Komme Rijn area (S22).

In our model, bee habitat quality and pollination potential increased by either: the presence of hedges, a transition to organic farming or by taking pastures out of production. Agri-environment measures in orchards and on pastures nearby orchards also increased the pollination potential within orchards.

We used a look-up table approach to quantify the profits of fruit yield per cell. We obtained production estimates, costs and market prices for organic and conventional orchards producing apples and pears (Heijerman-Peppelman and Roelofs, 2010; de Groot et al., 2015, 2016). Again we used a transition period to account for the costs associated with switching to organic farming. The transition period for fruit yield is three years (SKAL, 2017).

We calculated the total fruit yield over a period of ten years. Given that the current market prices of organic fruit tree production are high, the costs endured in the first three years are outweighed by the additional profits in the later years.

Aesthetic quality

We quantified the aesthetic quality of the landscape using a model specifically designed for the Kromme Rijn area (Tieskens et al., 2018). The model linked the amount of unique user uploads of landscape photos on social media platforms to the location of a set of structural landscape features (Panoramia and Flickr). Most spatial predictors in the model were relatively fixed, including the location of forts, rivers, castles, walking and bicycling roads as well as population density. For the natural landscape features, the distance to both natural grasslands as well as linear elements had a significant impact on the amount of unique user uploads (Tieskens et al., 2018). Therefore, these type of agri-environment measures improved the aesthetic quality. Pastures and orchards did not affect aesthetic quality, irrespective of the management type (S22).

Great crested newt occurrence

The Kromme Rijn area is a focal area for the protection and restoration of habitat for the great crested newt (Utrecht Province, 2017). The nature conservation plan specifically mentioned the need to manage the land for this particular species in our study area (Utrecht Province, 2017). As a biodiversity indicator, we therefore opted for a model on the habitat suitability (number of individuals/pond) for the great crested newt within our optimization approach (van Teeffelen et al., 2015b). We calculated the carrying capacity for great crested newt for individual ponds as a function of the location of ponds and the amount of suitable habitat in the vicinity of the pond (van Teeffelen et al., 2015b). The newt model by Van Teeffelen et al. (2015b) was developed and applied in the Baakse Beek, a Dutch agricultural landscape dominated by dairy farming. Ponds are considered important for newts, as they require ponds for reproduction. The landscape surrounding a pond is used for feeding, shelter and hibernation in the juvenile and adult stage (Griffiths, 1996; Müllner, 2001).

In our model, the amount of individuals each pond can support depended on the habitat suitability of the surrounding landscape. Habitat suitability was a function of the amount of forested habitat, natural grassland and hedgerows within the surroundings of a pond (Van Teeffelen et al. 2015b). Pastures and orchards were not suitable habitat irrespective of the management type. We obtained the location of ponds (Utrecht Province, 2017) and calculated the potential number of individuals per pond based on the habitat suitability of the LULM surrounding the pond (250 meter radius) (Van Teeffelen et al. 2015b).

6.2.3 Landscape optimization algorithm

Objective functions

We used a multi-objective optimization algorithm to optimize the allocation of LULM in the Kromme Rijn area. Our goal was to simultaneously:

- maximize yearly fruit yield profit in euro /year on the land currently under production - calculated for a ten year period, summed for all orchard cells in the area;
- maximize potential newt habitat - measured as newt individuals per pond, summed for all cells containing ponds in the area;
- maximize landscape aesthetics - measured as the number of unique user uploads of landscape photos to social media platforms, summed for all cells containing hike or bike paths;
- minimize the loss in pasture production - measured as euro/year for a ten year period, summed for all pasture cells in the area.

Optimization algorithm

Optimization in this approach resulted in the increase in the objective functions following changes in LULM applied to pastures and orchards. We used a recently developed landscape optimization tool Constrained Multi-objective Optimization of Land-use Allocation "CoMOLA" (Strauch, 2018), a Python environment that can be linked to user-specific models. CoMOLA utilizes the non-dominated sorting genetic algorithm II (NSGA-II) (Deb et al. 2002) and allows to consider land use change constraints.

Based on the original LULM map (starting solution) and pre-defined constraints (land use transition and area proportion rules) the genetic algorithm first created a set of different yet feasible LULM allocation maps. This set is called a population. Each map is called an individual and is encoded as a string of integers, a so-called genome. Using the environmental models described above, each individual was assigned fitness values representing the achieved values for the four objectives. The algorithm then applied a Pareto ranking for each individual based on its fitness values, archives best individuals and selects individuals for mating to generate a new (offspring) population. In mating, each offspring individual was generated by a random combination (crossover) of two genomes. The likelihood of mating increased for individuals with a higher rank. Additional random mutations increased the diversity of genomes to consider a wide range of LULM allocations. Mating can result into infeasible, i.e. constraint-violating, offspring individuals. The genomes of infeasible individuals were modified using a repair mechanism specifically developed for land allocation optimization. The whole procedure from fitness value calculation to offspring generation and genome repairing was repeated until a termination criteria is reached, e.g. a convergence criteria or a pre-defined number of generations. In this manner the algorithm approaches towards the Pareto front, which allocates LULM in a way that no increase of one objective was possible without decreasing another. The Pareto-front thus defined the optimal trade-off between the objectives. Although no optimization algorithm can guarantee to find optimal solutions, genetic algorithms are known to find at least close to optimum solutions (Lautenbach et al., 2013).

Allocation of agri-environment measures

To maximize the objective functions, we allocated different LULM to pastures and orchards. Pastures and orchards could be assigned a total of four LULM based on a combination of farm management (conventional and organic) and establishment of linear elements (on-farm measure). In addition, pastures can be taken out of production (off-farm measure). As pasture production areas are most commonly restored to natural grasslands, we used values for natural grassland to calculate the effect of pastures taken out production on objective functions. Orchards could not be taken out of production.

Farmers were the primary decision makers to assign LULM to pastures and orchards. Due to computational limitations, in our approach it was only possible to allocate LULM to aggregated decision units. Given our focus on farmers, we used farms as decision units. We allocated individual cells to farms based on crop type, delineation of fields (CBS, 2014) and the location of dairy and fruit orchards (Utrecht Province, 2011) (S21). The Kromme Rijn area included a total of 277 pasture farms and 141 fruit orchards. In our aggregation routine, all cells in a farm were assigned a single LULM type. The decision to implement organic or conventional management is often taken at the farm level given the need to fully separate production cycles (SKAL, 2017).

The establishment of linear elements can differ both between and within fields. However, the optimization algorithm only allowed allocation of elements to all cells in the farm, given computational limitations that do not allow for allocation of elements to individual cells. In the calculation of each objective we accounted for the difference between field interior and field edge. Linear elements were assigned to the full farm. However in a second step we combined the map of linear elements at the farm level with a binary map of field edges and field interiors at the cell level. Thus linear elements were only allocated on field edges within farms that adopted linear elements as farm management technique (S21).

Optimizing the allocation of LULM started from a starting solution, usually the original LULM map. However, due to computational limitations we simplified the original LULM map for its use as starting solution in a way that only a single LULM is assigned to each farm. Given that farm management is assigned on farm level and linear elements differ within farms, we generated a starting solution without linear elements. For the starting solution, we only assigned the LULM conventional or organic management without linear elements to each farm. In the LULM optimization linear elements could be assigned at the farm level.

6.2.4 Agri-environment experiments

We ran a total of three optimization experiments, differing in the type of agri-environment measures implemented: 1) *on-farm:* pastures and orchards can only be assigned a change in farm management and/or establishment of linear elements. This experiment quantified the optimal trade-off for agri-environment measures only focused on changing farm practices, 2) *off-farm:* pastures can only be taken out of production, with no change in orchards. This experiment quantified the optimal trade-off if agri-environment measures would only focus on taking land out of production. 3) *all options:* pastures and orchards can be assigned to all LULM categories, combining *on-farm* and *off-farm* agri-environment measures. This experiment quantified the optimal trade-off if all LULM options would be combined. Across all experiments the LULM allocation was performed for 277 pasture farms and 141 fruit orchards.

In addition to the LULM transition rules defined for each experiment, we limited the extent of change in allocation of LULM per category as a further optimization constraint. For pastures, we calculated the amount of land taken out of production proposed in the conservation plan and limited the amount of change from conventional management to another LULM allocation by this. Therewith we limited the loss in pasture production by the expected loss incurred in the nature conservation plan. We used yearly transition rates of conventional to organic orchards to calculate the allowed change from conventional orchards to orchards with agri-environment measures (S22).

Last, for each experiment we used a population size of 100 LULM allocations (individuals) and ran a total of 100 generations. We repeated each experiment ten times. For each experiment we thus generated a total of 10.000 LULM allocations for comparison in each repetition.

6.2.5 Analysis of results

Each landscape optimization experiment resulted in a set of non-dominated LULM allocations. For each land allocation we calculated a set of indicators. First, we visualised the functional trade-offs between objectives by calculating the percentage change for a LULM allocation per objective relative to the value of that objective for the starting solution. The starting solution was a simplified version of the current landscape without linear elements and management practices assigned at the farm level.

Second, any multi-objective landscape optimization will generate a diverse set of optimal outcomes. A challenge lies in translating a large set of non-dominated outcomes to targeted spatial planning advice. We adopted an approach by Karakostas (2017) and calculated the relative frequency a cell was assigned to agri-environment measures across all LULM allocations per experiment. This approach identified locations that (almost) always have agri-environment measures irrespective of their location at the Pareto-front.

Last, we compared the outcomes of the optimization analysis to the current landscape and the proposed conservation plan. For each LULM allocation we calculated the relative change compared to the current landscape, containing linear elements and varied LULM at the cell level. This analysis provides insight into the extent that the current landscape and the nature conservation plan can be improved with different LULM allocations.

We performed ten runs for each scenario. For each run, we calculated the hypervolume of the Pareto front as a performance measure indicating the likelihood that the Pareto front will move further outward, i.e. values per indicator might increase (Zitzler and Thiele, 1999; Jiang et al., 2014) (S23). For each scenario we selected the run with the best performance, i.e. highest hypervolume value at the end of the run. The similarity in the hypervolume curves among repetitions indicates an overall robustness of our results and is unlikely to change the overall outcomes. All analyses were performed for the non-dominated solutions for that run. All analyses were performed using R (R Core Team, 2016).

6.3 Results

6.3.1 Functional trade-offs per agri-environment measure

All agri-environment experiments increased the environmental objectives at the cost of pasture production (Figure 6.2). We visualized the four-dimensional Pareto-front for the *all options* experiment (Figure 6.2, top Panel). In addition we visualized the trade-off between pasture production and each environmental objective for *all options* (Figure 6.2 A), *off-farm measures* (Figure 6.2 B) and *on-farm measures* (Figure 6.2 C). The extent to which each environmental objective is enhanced differed strongly per experiment. Focusing solely on *on-farm* measures, these strongly enhanced fruit yield (max 26.19% increase), calculated as the change in fruit yield relative to the starting solution, but had a smaller (positive) impact on newt habitat (max 9.90% increase) (Figure 6.2 C).

In contrast, focusing on *off-farm* measures had a strong positive impact on newt habitat (max 45.72%), but hardly enhanced fruit yield (+1.63%) (Figure 6.2 B). Aesthetics was most strongly affected by *off-farm* measures, but the effect of all measures was relatively small. The limited gains of different LULM for landscape aesthetics was due to the fact that aesthetics is affected by many fixed factors, such as distance to castles, water and forest.

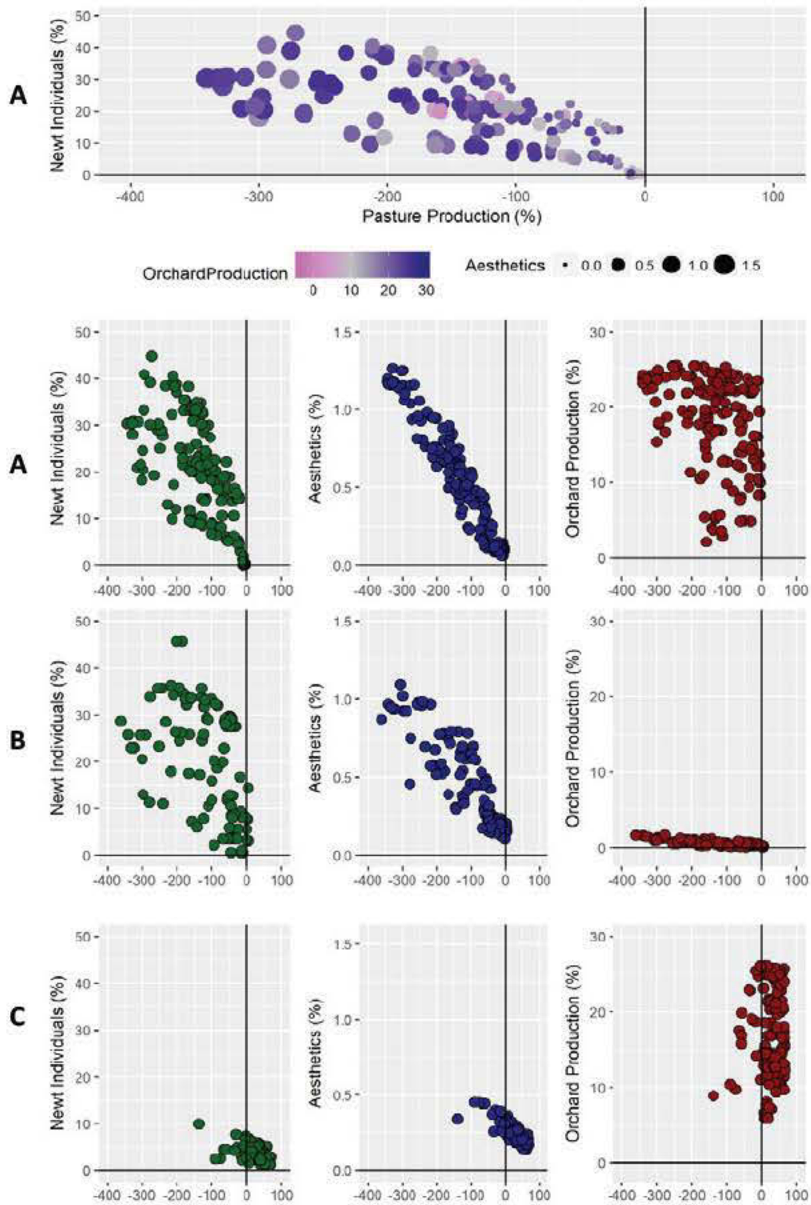
In general, decreasing pasture production resulted in increases for all other objectives. However, a 26.2% increase in fruit yield can also be achieved without losses in pasture production (Figure 6.2 C). These strong initial gains were achieved by *on-farm* measures on orchards. Increases in the other objectives can only be achieved with decreasing pasture production and thus always result in a trade-off.

Agri-environment measures including all LULM allocations, *all options*, was capable of simultaneously enhancing newt habitat and fruit yield (Figure 6.2 A, B). While the maximum values for fruit yield (+25.57%) and newt habitat (+44.72%) were slightly higher than the maximum values that could be achieved in the other experiments, these differences are very small. Overall, these results showed that all agri-environment alternatives considered can increase the landscape's capacity for these three environmental objectives simultaneously, demonstrating how the current landscape can be enhanced. Having said that, the strength of the trade-offs among objectives changed with different sets of agri-environment measures available, indicating that careful target-setting is important.

6.3.2 Spatial priorities for agri-environment measures

We identified spatial priorities for LULM allocations (Figure 6.3). We only visualised the results for the *all options* experiment, because this experiment has the highest potential for creating synergies among objectives. In this experiment, only a few areas were assigned an agri-environment measure across all optimal LULM allocations (Figure 6.3 A). These areas were mainly located in and around orchards in the centre of the Kromme Rijn. This showed a mismatch with the areas designated in the nature conservation plan, as these are mostly at the edge of the Kromme Rijn area, for example around rivers and existing nature areas (Figure 6.3 D). The conservation plan is solely developed for species protection, but does addresses more

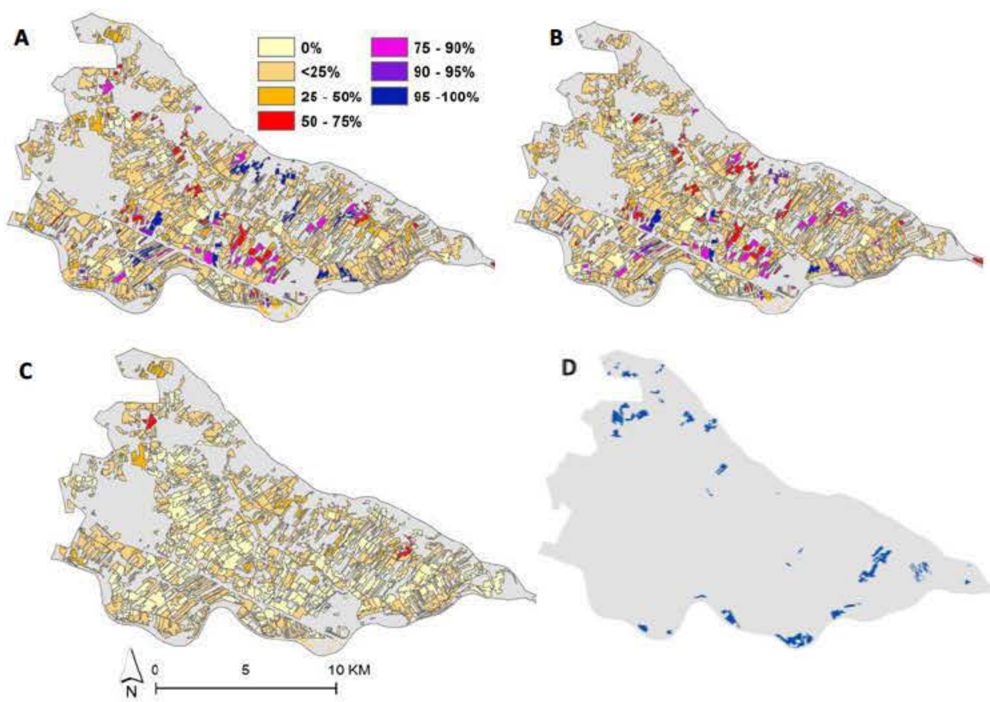
Figure 6.2: Optimal land use land management allocations per agri-enviroment experiment. Each point represents a single optimal solution. The top figure represents the 4-dimensional trade-off curve for the "all options" experiment (A). The bottom nine panels represent the trade-off between pasture production and each environmental objective for (A) all options, (B) off-farm measures and (C) on-farm measures. Pasture production can decrease by more than 100% because it is quantified as an opportunity cost (i.e. a loss) and not asthe actual profit from pasture production. The interested reader can also find a 3d visualization of the pareto frontier for each scenario in the S21.



species than just the newt, partly explaining the difference in locations. Our results indicate that if additional objectives and agri-environment measures would be included in the conservation plan, the preferred locations for these measures would be located differently. Mainly, preferred locations would move to the centre of the Kromme Rijn area in and around orchards.

The frequency with which a cell is assigned to an agri-environment measure does not indicate the type assigned to that cell. Therefore we separately identified the frequency of *on-farm* (Figure 6.3B) and *off-farm* measures per cell (Figure 6.3C). Specific locations have a high frequency of *on-farm* measures, especially in and around orchards. These locations are thus identified, across the different runs, as priority areas for implementing both agri-environment measures, but more specifically for implementing *on-farm* measures. Interpretation should be done carefully as the locations embed different trade-offs. The frequency for *off-farm* measures per location was far lower (Figure 6.3C) indicating that no location is clearly a priority to be taken out of production. Thus although a combination of both *on-farm* and *off-farm* measures is preferred, only priority locations for *on-farm* measures could be identified.

Figure 6.3: Priority map for agri-environment measures in the Kromme Rijn area. The higher the priority the more often a field is assigned to a agri-environment measure. Results are depicted for *all options,* i.e. a combination of both *on-farm* and *off-farm* measures. The maps depict priorities for (A) conventional versus all other agri-environment measures, (B) *on-farm measures* versus all other, (C) *off-farm* measures versus all other and (D) the areas targeted in the nature conservation plan to be taken out of production (blue areas in D).



6.3.3 Comparison to current and regional policy

The conservation plan resulted in a 15.54% increase in newt habitat compared to the current landscape, at the cost of a 37.21% decrease in pasture production (Figure 6.4). The conservation plan is primarily designed for species protection and the increase in newt habitat highlights that the plan is relatively effective in this regard. However, the conservation plan addresses more species beyond the newt which we cannot incorporate in our comparison given the model limitations. For the other objectives the conservation plan did not improve the aesthetic value of the landscape (0%) and decreases fruit yield (-19.35%). Fruit yield strongly declined because a small set of orchard fields is taken out of production in the current conservation plan.

We compared both the current landscape and conservation plan to the results from the optimization analysis. One has to be aware that the landscapes were somewhat different because the current and nature conservation plan allocates measures at the cell level whereas the optimization analysis has homogeneous LULM (allocation) at the farm level. Nonetheless both the current landscape and the conservation plan could be further optimized for all four objectives (Figure 6.4). For the current landscape there were multiple Pareto-optimal land allocations that performed better on the environmental objectives, without reducing pasture production. For example, compared to the current landscape a different LULM allocation could increase newt habitat (+22%) for fewer losses in pasture production (+22%) and simultaneously increase the other environmental objectives.

For the conservation plan there were also multiple Pareto-optimal land allocations that performed better on the environmental objectives, without reducing pasture production. Compared to the conservation plan a different land allocation resulted in a smaller loss in pasture production (-37.21% vs -34.74%) and additional increases for all three environmental objectives (orchard: +22.94%, aesthetics: +0.83%, newt: + 25.85%). These results highlighted that combining *on-farm* and *off-farm* measures can further increase newt habitat with smaller losses in pasture production compared to the conservation plan. In addition, fruit yield and landscape aesthetics could be increased alongside newt habitat in the Kromme Rijn Area.

6

6.4 Discussion

6.4.1 Reflection on the main results

A landscape optimization for multiple objectives is capable of identifying functional trade-offs between competing objectives (Lautenbach et al., 2013). Previous analyses optimized the landscape allocating multiple management options but did not break the separate effects of each option apart (Lautenbach et al., 2013; Pennington et al., 2017).

As a result, we found that all agri-environment measures could improve all environmental objectives simultaneously, but that the improvement of each objective differed between *on-farm* and *off-farm* measures. Either choosing a management strategy focused on "*off farm*" or "*on farm*" resulted in a strong trade-off between newt habitat and fruit yield. Allowing a mixture of measures could alleviate this trade-off. This provides important information for landscape planners beyond identifying the functional trade-off for an allocation of all LULM options to navigate trade-offs based on the agri-environment measures implemented.

All three environmental objectives were sensitive to landscape configuration including functions related to surrounding landscape (distance decay), edge effects and linear elements. The importance of landscape configuration for ecosystem services has been previously addressed using a conceptual model (Mitchell et al., 2015a) or model comparisons (Lautenbach et al., 2011; Verhagen et al., 2016). The Pareto frontier in any optimization analysis can be interpreted as the set of optimal landscape configurations for increasing opportunity costs. It highlights what can be optimally achieved when combining the effect of the area of interventions (composition) with the spatial allocation (configuration). In addition, our results highlighted the importance of linear elements in agricultural landscapes for ecosystem service supply. Increases in newt habitat and aesthetic quality of the on-farm experiment can be solely attributed to increasing the amount of linear elements given that organic farming has no impact on these objectives in our models. Previous research linked these elements to a diverse set of ecosystem services and argued for the inclusion of elements in landscape optimization approaches (Jones et al., 2013; Verhagen et al., 2016). To our knowledge this is the first application of linear elements in landscape optimization. More importantly, we demonstrated that a landscape optimization without these elements would result in a less optimal outcome (Figure 6.2), thus providing evidence for the importance of accounting for configuration and linear elements in landscape planning.

In this paper, we attempted to bridge the gap between multi-objective optimization resulting in numerous alternative land allocations and specific spatial planning recommendations. Any multi-objective optimization will result in a large set of Pareto dominant solutions (Karakostas, 2017) and previous research has translated these numerous outcomes into spatial planning recommendations by providing example landscapes along the Pareto front (Nelson et al., 2009; Gourevitch et al., 2016; Kennedy et al., 2016; Pennington et al., 2017). A method to effectively visualize optimization outcomes can help to discuss policy targets (Verburg et al., 2016). Based on additional constraints and discussions with stakeholders, one could potentially arrive at a single optimal LULM allocation. However, such an approach is criticized for not showing the pathways to reach such outcomes (Verburg et al., 2016). Here, we have aggregated all optimization outcomes to identify priority areas for agri-environment measures, following an approach by Karakostas (2017). By doing so, we could move beyond visualizing exemplar landscapes, by using all optimization results to inform spatial planning more directly by delineating target areas suitable for successful policy implementation, which can subsequently be used in policy design and implementation. Furthermore, our approach is suitable to inform policy design directly, as the impact from different policy mixes is explicitly assessed.

Condensing the full set of pareto optimal solutions into a single spatial map risks ignoring important trade-offs and preferences between the solutions. The involvement of relevant stakeholders before (*a priori*), during (interactive) or after (*a posteriori*) the optimization process is vital, to understand the local decision making context, as well as to provide a suitable fit with local interests and ecosystem services demand (e.g. Bryan et al. 2010; Cord et al. 2017). In addition, understanding the needs of planners and policy makers is also an important factor for a successful policy uptake of this type of decision-support tools (e.g. McIntosh et al. 2011; Albert et al. 2014). Therefore identification of priority areas for restoration has the potential to inform policy makers and stakeholders by providing spatial explicit information on the location of restoration measures but should not be easily used as a technical replacement of stakeholder deliberation as tradeoffs may be judged in different ways by different stakeholders and other considerations, not accounted for in the optimization, may play an important role. One step to better include stakeholder deliberation in the approach is to assign relative weights to each outcome, instead of assuming an equal weight for all alternatives, as it is done in our identification of priority locations. Therefore, care should be taken in not interpreting the highest priority areas for restoration as a single best solution but as a way of presenting optimization results in a spatially explicit manner and to inform and not end a discussion among stakeholders.

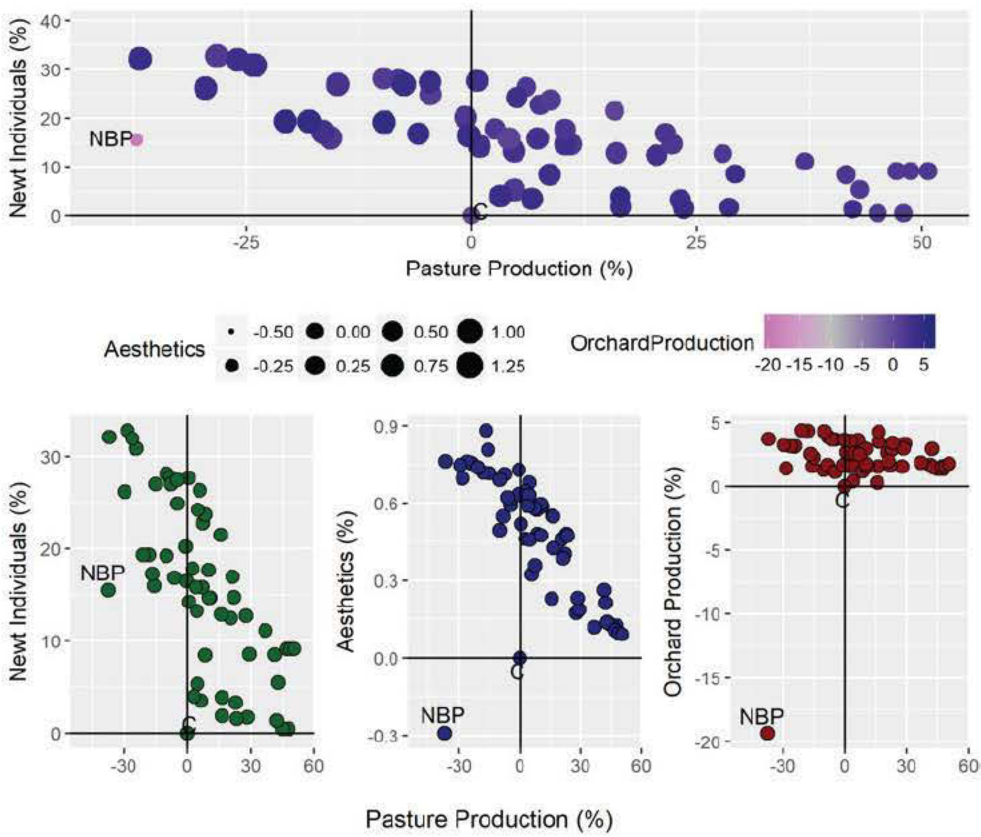
6.4.2 Limitations to our approach

Any environmental model and computational optimization requires a set of simplifications and assumptions. In our case, a limitation is the necessity to implement LULM alternatives at the farm level. This assumes a uniform LULM across all cells in a farm. However, in reality many agri-environment measures can be implemented below the farm level. Therefore, the resolution at which the optimization model and the landscape planner allocate

LULM is not aligned. A clear example of this is the establishment of linear elements, which in reality does not necessarily occur on all fields of a farm. We partly accounted for this by only allowing allocation of linear elements on field edges.

An importation decision is the use of the optimization algorithm, in our case NSGA-II. Several optimization algorithms exist that can be used for multi-objective optimization such as BORG, SPEA-2 or NSGA-III (Deb and Jain, 2013; Hadka and Reed, 2015; Cholodowicz and Orlowski, 2017). Here we choose to use the NSGA-II given its' wide applicability in spatial allocation problems (Malczewski and Rinner, 2015). Our analysis focuses on the use of optimization algorithms in a real world case study and policy framework. A methodological comparison of individual optimization algorithms is outside the scope of this paper. Importantly, using NSGA-II algorithm we can highlight that both the current situation and the nature management plan can be optimized for multiple objectives in our study area.

Figure 6.4: Set of Pareto dominated solutions from the optimization analysis. All changes are calculated relative to the current landscape. The solutions to the top right of the current landscape (C or 0,0) are Pareto dominant, i.e. score better on all objectives compared to the current landscape. The nature conservation plan (NBP) is Pareto dominated, score better on all objectives, by all solutions to the top right of the NBP dot. The results depicted are the outcomes of the "all options" experiment.



Any optimization model depends on the quality of the underlying models used to calculate the individual objectives. A first clear limitation of our approach is that we used a single species model as a proxy for biodiversity. The newt is an important focal species for our study area and much effort goes into habitat protection and restoration for this particular species. Therefore, it has a direct policy relevance for our study area. However, the great crested newt should not be interpreted as a proxy for overall response of biodiversity, given its strict reliance on the combination on habitat restoration in the vicinity of ponds. A second limitation is that we used average production values to quantify both pasture and orchard production. For orchard production the production values are based on local case studies in similar regions. For pasture production we used national averages given a lack of more specialized data. Given the relatively small extent of the study area and the similarities in management intensity across the region a single average pasture production value was deemed a valid assumption. A third limitation is that we used social media pictures as a proxy for aesthetic quality. The use of this proxy might introduce a bias to specific users of the area as well as to specific aesthetic qualities particularly suitable to capture through photos. The major advantage of this model is that it was specifically developed for this study region (Tieskens et al., 2018).

6.4.3 Management recommendations

The current nature conservation plan focuses on taking designated areas out of production to meet species conservation objectives. Here we showed that with a more optimal allocation of measures, it was possible to meet species conservation objectives while simultaneously improving on a number of other environmental objectives. Moreover, we showed that the addition of *on-farm* measures in combination with *off-farm* measures is more effective in realising the objectives. The nature conservation plan combines taking designated areas out of production with promoting voluntary measures on agricultural land. We showed that explicitly incorporating *on-farm* measures would be a more effective strategy. Focusing on establishment of linear elements and organic management might be more easily implemented because it involves lower costs to farmers, as well as the possibility to obtain financial support for establishing linear elements.

In addition, our spatial analyses showed that priority areas for agri-environment measures do not align with currently designated areas. These results can assist spatial planners in designing future plans. The current management strategy focuses on farms forming voluntary collectives proposing to implement agri-environment measures. Spatial explicit knowledge on the location of priority locations for restoration can help evaluate the effectiveness of these proposal.

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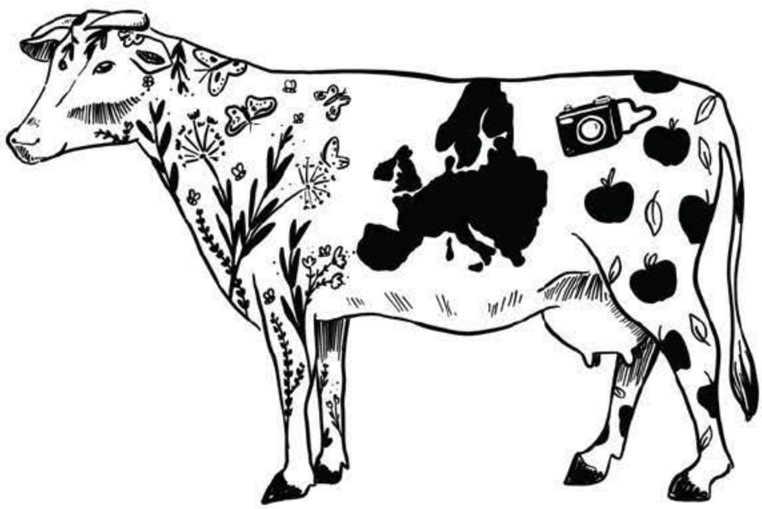


7. Synthesis and outlook

7.1 Objective

The overall objective of this thesis is to develop and apply methods for integrating multiple ESs into landscape prioritization and optimization tools. These tools are applied to identify priority areas and actions for the conservation and restoration of land for multiple ESs. Specifically, the thesis focused on the role of land cover change, landscape configuration and land management in a context of societal demands for these services. Throughout this thesis I focused on multiple ESs and addressed how to minimize and navigate trade-offs between ESs. I addressed the following research questions:

(RQ1) How does the spatial configuration of land cover affect the capacity of the landscape to provide ESs, and what are the implications thereof for mapping ESs?

(RQ2) How are the trade-offs between biodiversity and ESs affected by land management intensity?

(RQ3) How do societal demands for ESs, landscape configuration and management intensity change the relative priority of areas for ESs?

(RQ4) What are the opportunities in multifunctional landscapes for restoration through changes in land cover, landscape configuration and management intensity?

In this chapter I provide an overview of the main findings related to these research questions, limitations related to the research in this thesis and the implications for future scientific work. After this overview, I discuss the policy implications of the main findings.

7.2 Landscape configuration and quantifying ES capacity

For a subset of ESs, landscape configuration affects the capacity of the landscape to provide ESs (Ch. 2). The effect of landscape configuration on these ESs is not uniform but depends on the ES studied. We identified four different ways in which landscape configuration can affect ES capacity. These four ways are i) distance to fixed objects, ii) configuration of a single patch, iii) configuration of multiple patches and iv) landscape elements. Each ES capacity was affected differently. For example, flood regulation was mostly affected by distance to nearby streams (i) and the presence of linear elements (iv), whereas landscape aesthetics was affected by the interaction of multiple land cover patches (iii). Thus, whether and how landscape configuration affects ESs depends on the ES studied, the landscape configuration effect considered and the local landscape conditions.

7.2.1 Integrating configuration in ESs mapping approaches

Accounting for landscape configuration in mapping ESs resulted in strong local variation in ES capacity (Ch. 2). We made a comparison between two approaches that did or did not account for landscape configuration in the mapping of ESs in Scotland. Accounting for landscape configuration in mapping ESs changed the level and spatial variation of ES capacity. The differences were strongest at the local scale. Across larger scales (watershed or national) the local variation in ES capacity following landscape configuration mostly averaged out.

Effects of landscape configuration are not systematically integrated in the mapping of ESs. The mapping of ESs still heavily relies on land cover proxies and look-up tables (Seppelt et al., 2011a; Martínez-Harms and Balvanera, 2012). However, in recent years some progress has been made. InVEST is a widely applied tool for the mapping of ESs. It now integrates effects of landscape configuration in the quantification of pollination, erosion control and flood regulation. Recent developments of InVEST include a module carbon storage, with variation between forest edges and forest interiors (Sharp et al., 2015). This module is based on findings that carbon storage in the tropics is on average 25% reduced in forest edges compared to interiors (Chaplin-Kramer et al., 2015). This is a promising example of mainstreaming effects of landscape configuration in tools to map ESs.

Local effects of landscape configuration on ESs are especially relevant to identify priorities for managing land for ESs. Variation in ES capacity as a result of landscape configuration is particularly strong at the local scale. Therefore, assessments of ESs across larger extents and coarser resolution do not necessarily need to account for configuration when quantifying ES capacity. Especially if these studies aim at a first broad estimate of ES levels. In such cases, the spatial variation within the area is of lower interest. However, studies that focus on smaller spatial extents or spatial variation in ESs do need to consider configuration. This spatial variation is important to identify where to prioritize land for ESs. In other words, maps produced to provide a first estimate of ES capacity are not appropriate to identify priority areas for ESs. The strong local variation in ES capacity due to landscape configuration is pivotal to identify priority areas for ESs.

7.2.2 Improving the evidence base

In the following paragraphs I discuss the limitations of our study and some recent advances in linking landscape configuration and ESs. The focus is on three specific topics. In the first paragraphs I discuss the evidence base, the following paragraphs centres on how landscape configuration affects ES capacity and ES flow. In the last paragraphs I discuss the role of landscape elements .

We performed a literature review on effects of landscape configuration on multiple ESs. Although this review provides evidence for the importance of landscape configuration to assess ESs, I conclude that the evidence base needs further strengthening. The evidence base can be further improved through field studies and meta-analysis. The number of studies that looked at a relation between landscape configuration and ESs strongly differed per ES. A previous review, focusing on the relation between fragmentation and ESs, only retrieved 15 studies (Mitchell et al., 2013). We found considerably more studies. Likely, because we included multiple effects of configuration besides fragmentation and included studies that did not mention the term “ESs”. Still the overall amount of studies remains limited and is strongly biased towards a small selection of ESs. For ESs that are strongly related to the field of ecology and species movement (i.e. pollination and pest control) many field studies as well as meta-analyses are available. For other services either field studies or meta-analyses do not exist. A possible explanation is that research fields on other services, like hydrology or soil science, are not well connected to the field of ESs and landscape ecology. This limits the uptake of the findings. A next step could be to setup specialized meta-studies for research on effects of landscape configuration on single ESs.

The evidence base could also be improved by studying effects of landscape configuration on multiple ESs simultaneously (Seppelt et al., 2011a; Mitchell et al., 2013). This limits our knowledge on how landscape configuration affects interactions among multiple ESs (Bennett et al., 2009; Raudsepp-Hearne et al., 2010; Renard et al., 2015). In this thesis, I specifically focus on managing the land for multiple ESs. However, many of the effects of landscape configuration on ESs are based on studies looking at ESs in isolation. Therefore, how interactions and trade-offs between services are affected by landscape configuration remains largely unknown (but see Mitchell et al., 2014a). More field research is needed to further expand knowledge on how landscape configuration affects multiple services simultaneously.

Future field studies and reviews should prioritize studies on the effects of landscape configuration on flood regulation and pest control. Our review showed that for these services the effects of landscape configuration were inconclusive. A recent meta-study further looked at effects of landscape configuration on pest control (Karp et al., 2018). They concluded that there was insufficient evidence that increases in natural vegetation around agricultural fields support natural pest control and reduce crop damage (Karp et al., 2018). He finding that no generalizable effect exist is largely in line with our findings. A next step would be to research under which conditions landscape configuration does or does not affect pest control. Similar research could be performed for flood regulation. For other services there is a need to expand the amount of field studies prior to conducting reviews on generalizable effects of landscape configuration. To conclude, there is a need to expand the evidence base through more field studies, especially studies that look at effects of landscape configuration on multiple ESs simultaneously. There are relatively many studies on flood regulation and pest control and therefore these ESs should be prioritized for reviews and meta-analyses For many other ESs more field studies are required prior to conducting reviews on effects of landscape configuration on ESs.

7.2.3 Effects of landscape configuration on ES flow and ES capacity

Landscape configuration impacts both ES capacity (Ch. 2) and ES flow (Ch. 4). Chapter 2 primarily focused on the effect of landscape configuration on ES capacity. In Chapter 4 we focused on the configuration of areas of ES demand and ES capacity, the ES flow. We showed that accounting for ES flow resulted in a strong change in the relative priority ranking of areas.

Previous research has argued that fragmentation negatively affects ES capacity but positively impacts ES flow (Mitchell et al., 2015b). There is debate on the validity of this assumption (Andrieu et al., 2015). Results from literature and this thesis show that fragmentation can have positive and negative impacts on ES capacity. In general, fragmentation reduces carbon stocks in tropical forest edges (Chaplin-Kramer et al., 2015). However, fragmentation and increased edge density can increase habitat suitability for wild bee pollinators (Ch. 2). Similar contrasting effects have been found for biodiversity. A literature review reports both positive and negative effects of fragmentation on biodiversity, with the majority actually reporting positive impacts of fragmentation on biodiversity (Fahrig, 2003, 2017).

There is much less research available on the impact of fragmentation on ES flow. In general, one can conclude that effects of fragmentation on ES flow also depend on the ES studied (Ch. 4). ES flow is strongly governed by the spatial relation between ES demand and ES capacity, or in essence the configuration of these two (Schröter et al., 2014a; Serna-Chavez et al., 2014). These relations are in some cases omnidirectional and not spatially restricted, where the beneficiary can benefit from the ES capacity at a site irrespective of the distance (e.g. climate regulation through carbon sequestration). In other instances connections are strictly in situ, where the beneficiary can only benefit from an ES capacity if the beneficiary is present at that specific location (Serna-Chavez et al., 2014) (e.g. noise attenuation). The more spatially restricted the ES flow, the more important the configuration of ES demand and capacity areas. For a omnidirectional service like carbon sequestration changes to ES flow following fragmentation are completely governed by changes in ES capacity. Fragmentation can increase the accessibility of a site, or more in general can increase the connections between areas of ES demand and ES capacity (Mitchell et al., 2015b). Fragmentation could thus especially be beneficial for local and regional flow ESs. However, this argument does not hold for global flow ESs. Carbon sequestration, in the tropics, is reduced by fragmentation (Chaplin-Kramer et al., 2015). Logically, it follows that also ES flow of carbon sequestration is reduced by fragmentation. Thus, there is no real evidence to support a claim that fragmentation in general improves ES flow.

To conclude, ES flow and ES capacity are separately and independently affected by changes in landscape configuration. Based on the literature and the findings in this thesis I thus conclude that ES flow and ES capacity are both affected by the configuration of landscapes, though not necessarily in similar ways. Landscape configuration uniquely affects each ES, and uniquely affects the ES flow and ES capacity of each service. There is thus no support for generalizable positive and negative effects. Therefore, future research needs to independently assess effects of configuration on ES capacity and ES flow for individual services.

7.2.4 Landscape elements and ES capacity

Landscape elements positively impact ES capacity of several services (Ch. 2). Landscape elements can prevent and intercept the flow of water, nutrients and sediment, can provide habitat to species beneficial to agricultural production and are in some landscapes important parts of the cultural agricultural identity. Previous research also argued that restoring landscape elements can be a crucial component of creating more multifunctional agricultural landscapes (Jones et al., 2013). We integrated landscape elements in the identification of priority areas (Ch. 4) and actions (Ch. 6).

However, most studies mapping ESs hardly account for landscape elements. Landscape elements are difficult to detect in conventional land cover data sets. Nowadays, maps of the distribution of hedges, grass margins and stone walls across the EU exist (van der Zanden et al., 2013). Recent ES maps of pollination and erosion control for the EU have integrated the distribution of landscape elements in their mapping approach (Schulp et al., 2014b; Panagos et al., 2015). Based on our review and the importance of landscape elements for prioritization, I conclude that future studies mapping ESs should aim to incorporate landscape elements in mapping ESs.

So far we are missing scenarios or data on the future extent of landscape elements. Therefore, we could not integrate landscape elements in all chapters of this thesis. (Ch. 5). Developing scenarios on the future extent of landscape elements at the EU level can provide insight in the future provision of multiple ESs and support policy assessments such as those on no net loss (Schulp et al., 2016) or guide the EU-wide strategy promoting investments in green infrastructure (Liquete et al., 2015; Maes et al., 2015a). Therefore, a next step is to incorporate changes in the distribution of landscape elements into land use change scenarios.

7.3 Land management and ESs

In a meta-analysis (Ch. 3), we found a general trade-off between biodiversity and production following management intensification. Management intensification had significant negative effects on biodiversity (-8.9%) and significant positive effects on production (+20.3%). However, between studies there was large heterogeneity in the response of production and species richness to intensification. Two factors explaining this heterogeneity were species groups and production systems (cropland, grassland and forest). Only in croplands a significant trade-off was observed resulting in decreased species richness (-21.2%) and increased production (+33.3%). In forest and grassland systems either production or species richness did not have a significant response to intensification. Effects of intensification also differed between aggregate species groups. Plant species richness decreased significantly following intensification, whereas animal species richness was not significantly affected. Thus, management intensification resulted in a general trade-off between biodiversity and yield, but production systems and species groups governed the strength of the trade-off.

The strength of the change in intensification also affected the strength of the trade-off. We distinguished four levels (low, medium, high) and six steps in intensification. These steps are a combination of the initial intensification level (low, medium, high) and the final intensification level (low medium high). Only intensification within medium intensity systems (initial: medium-final: medium) resulted in a significant trade-off between biodiversity and yield. For other intensification steps we found a win-no loss response, i.e. intensification resulted in significant gains in production, without significant decreases in biodiversity. None of the intensification steps resulted in a win-win, i.e. significant gains in biodiversity and production following intensification. In the next section I discuss how to further improve the evidence base (7.3.1) and how management intensity affects biodiversity and ESs (7.3.2).

7.3.1 Providing a solid evidence base

The question how to simultaneously provide sustainable food production and conserve biodiversity s heavily debated in the scientific literature (Green et al., 2005; Phalan et al., 2011b; Tscharntke et al., 2012; Fischer et al., 2014; Seppelt et al., 2016). Using a meta-analysis approach we found a significant trade-off between biodiversity and crop production. This result further emphasizes the need to reconcile food production and biodiversity and to find alternative pathways of sustainable intensification that can do so (Fischer et al., 2008; Clough et al., 2011; Tilman et al., 2011; Garnett et al., 2013; Scherer et al., 2018). However, we also found that there is a lack of field studies looking at the effects of intensification on yield and biodiversity simultaneously. In contrast, many studies looked at effects of intensification on biodiversity or production separately. To effectively address questions on reconciling food production and biodiversity, field studies need to cross divides between research disciplines. Studies that address trade-offs between multiple objectives are of high relevance for global sustainability.

Previous research has argued that adoption of different management techniques in agriculture might hold the potential to generate win-wins between biodiversity, ESs and yield over time (Palm et al., 2014; Smith et al., 2016). Except for some specific case studies, our results do not support the existence of generalizable win-wins. This does not necessarily suggest that changes in management intensity cannot benefit both production and biodiversity. Our database consisted of a wide variety of management options. Nonetheless, these options mostly look at intensification in industrial agriculture and less so at sustainable intensification (Struik and Kuyper, 2017). More specialized reviews on either specific management options or specific production systems might find support for so-called win-wins.

For specifics steps in intensification we did find win-no loss relationship. Win-no loss relations represent significant gains in production without significant changes to biodiversity. This is a promising result suggesting that under specific circumstances no trade-off occurs between production and biodiversity. However, care should be taken in interpreting this result. A limitation is that our study looked at species richness as an indicator of biodiversity. Previous research has questioned the extent to which species richness metrics can adequately capture changes in biodiversity. A meta-analysis on biodiversity-intensification relationships highlighted negative relationships for both species richness and other metrics of biodiversity (Newbold et al., 2015). They concluded that after land use, land management intensity most strongly influenced species richness and local abundance (Newbold et al., 2015). However, other studies have highlighted contrasting responses of species richness and abundance metrics, concluding that solely reporting on species richness potentially masks effects of intensification on biodiversity beyond species richness (Gonzalez et al., 2016; Hillebrand et al., 2018). So it is questionable whether responses in species richness can be directly generalized towards other components of biological diversity. The majority of case studies in our database reported on species richness, thus limiting our options to include species composition and abundance metrics alongside species richness. Therefore, additional research is required to test the validity of win-no loss relationship for multiple indicators of biodiversity. To be able to do so case studies need to report on metrics of biodiversity beyond species richness. Only then general conclusions can be drawn about potential win-no loss relationships for intensification, production and species diversity.

7.3.2 Management intensification and ecosystem services

Our meta-analysis (Ch. 3) focused on provisioning ESs and biodiversity. In contrast to other chapters, we did not incorporate regulating or cultural services. Even though chapters 5 and 6, like chapter 3, dealt with land management intensity, it is not possible to transfer results on the effects of intensification on biodiversity (Ch. 3) directly to regulating and cultural ESs.

Previous research found support for linkages between changes in biodiversity and regulating services following intensification. In general, studies have found positive links between biodiversity levels and ES capacity (Naeem and Li, 1997; Cardinale et al., 2012; Balvanera et al., 2014). Although uncertain and likely to depend on local landscape conditions, losses in biodiversity following management intensification are likely to negatively affect regulating and cultural ESs (Cardinale et al., 2012; Mace et al., 2012). Specifically for intensification, studies found that management intensity affects ESs both directly and indirectly through changes in biodiversity (Allan et al., 2015; Winter et al., 2018). Therefore, there is some support that declines in biodiversity following management intensification are likely to negatively affect regulating and cultural services.

Further intensification of food, grass and wood production is important to meet current and future demands for provisioning services (Mueller et al., 2012; Kuemmerle et al., 2013; Verburg et al., 2013; van Ittersum et al., 2016; Erb et al., 2017; Wu et al., 2018). However, many production landscapes also provide and rely on regulating and cultural services. The interplay between intensification, production and multiple other services within a landscape requires more study. Studies addressing how intensification impacts regulating ESs at local and global scales are of critical importance to further inform sustainable intensification pathways (Tscharntke et al., 2012).

7.4 Identifying priority areas for ESs

In chapters 4 and 5 we identified priority areas for ESs in Europe, whilst accounting for societal demand (Ch. 4) and land use change (Ch. 5). We identified priority areas using ES maps that quantified these services based on effects of landscape configuration and management intensity. The focus in the second part of the thesis is on integrating ESs into prioritization tools. In the next sections, I discuss how integrating ESs into prioritization tools requires consideration of societal demand (7.4.1 & 7.4.2) and land use change (7.4.3). Next I specifically discuss how accounting for management intensity and configuration affect identification of priority areas for ESs (7.4.4).

7.4.1 Societal demand for ESs

For the first time, we identified priority areas for multiple ESs across the spatial extent of the EU. We identified priority areas based on ES capacity and based on a combination of ES capacity and demand. Accounting for demand for ESs made a large difference in the relative priority assigned to areas and on the amount of ES harboured by the top priority areas. Only after accounting for ES demand we can identify priority areas that harboured a much stronger direct benefit to human society (Ch. 4). In other words, accounting for demand resulted in different priority areas and higher levels of ES with relevance to society. Previous studies did not always use demand for ESs in the identification of priority areas. Based on the results presented in chapter 4, I conclude that not accounting for societal demand in identifying priority areas for ESs runs the risk of prioritizing areas with high ES capacity but limited benefits to human society. Therefore, accounting for ES demand is crucial in the identification of priority areas for ESs.

Accounting for ES demand shifts the focus from prioritizing well-functioning natural ecosystems to prioritizing areas with high benefits of nature to humans. This is not without risks. The prioritization analysis based on ES capacity yielded primarily remote natural areas as top priority areas. An analysis considering ES demand changed these priorities in favour of areas located nearer to cities and agricultural areas. In these areas people can benefit from localized services like air purification, flood regulation and pollination. However, a strong focus on human demand in the identification of priority areas might not sustain ESs and biodiversity in the long run.

There is discussion to what extent priority areas and conservation objectives for biodiversity and ESs align. Partly this discussion depends on scale. At larger scales high levels of ES supply are located within global biodiversity hotspots (Turner et al., 2007; Larsen et al., 2011, 2012). At smaller spatial scales there might be limited spatial congruence between top priority sites for biodiversity and ESs (Nelson et al., 2008; Paoli et al., 2010). Partly, this discussion depends on meeting multiple objectives. Adding more objectives almost always reduces performance for the initial objectives. Priority areas for biodiversity show mediocre performance for ESs, whereas priority areas for ES show mediocre performance for biodiversity conservation (Egoh et al., 2010; Reyers et al., 2012; Cimon-Morin et al., 2013). Identifying priority areas considering both objectives might partly overcome this issue but may still result in reduced conservation values (Chan et al., 2006; Larsen et al., 2011). Therefore, a strong focus on ES prioritization might further stretch already limited funds for conservation of biodiversity with limited potential to halt global biodiversity declines (Wilson et al., 2006; Steffen et al., 2015). This is a general issue in aligning conservation for both ESs and biodiversity. However, this issue might be further aggravated with a stronger focus on ES demand.

Identification of priority areas for local flow ESs is most affected by considering ES demand. Most of the spatial overlap between priority sites has been tested for biodiversity and carbon sequestration, a global flow ESs (Strassburg et al., 2010; Carwardine et al., 2015; Ferreira et al., 2018). However, the majority of ESs have a localized to regional ES flow (Chan et al., 2006; Cimon-Morin et al., 2013). Prioritization for localized ES flow is more strongly driven by demand side factors than by the actual biophysical properties of a site (Chan et al., 2006). Logically this shifts priority areas towards locations accessible by people. This particular shift creates two important risks.

First, current areas protected for biodiversity are disproportionally often located in remote, inaccessible locations to minimize human pressures on the landscape (Joppa and Pfaff, 2009). Current protected areas in Europe are relatively effective at the conservation of species (Kukkala et al., 2016). However, they are not necessarily effective at the conservation of ESs (Spanò et al., 2017). Emphasizing local flow ESs might shift priority areas away from biodiversity-rich but inaccessible locations.

Second, many ESs require a minimum use of the area to obtain benefits (Schröter et al., 2014b). This minimal use might negatively impact biodiversity conservation. In Europe the greatest threat identified to existing protected areas for biodiversity is recreational use (Schulze et al., 2018). Accounting for local and regional flow of ESs might further increase human pressures on priority areas for ESs and biodiversity. In the long term this could threaten the sustainable supply of ESs and biodiversity alike.

To conclude, accounting for societal demand has a strong impact on the identification of priority areas for ESs. Accounting for societal demand is a crucial component of identifying areas with a high level of ESs and with high benefits to society (Chan et al., 2006; Luck et al., 2012). However, accounting for ESs demand might put further strain on limited funds for the conservation of biodiversity. It might also jeopardize the conservation of areas with high biodiversity values and high ES capacity for societies in the future. Our results highlight that prioritizing land to meet societal demands for multiple ESs results in a shift in priority location. More importantly prioritizing land for ESs also shifts the focus towards prioritizing land with high benefits to human society. Therefore, it is crucial to include ES demand in the identification of priority areas. However, it is also crucial to strike a balance between identifying priorities for human benefits from nature and the intrinsic value of nature for current and future generations.

7.4.2 Accounting for distribution of priority areas across flow zones

Accounting for ES demand also means accounting for who is benefiting. We developed a novel method to account for the spatial distribution of beneficiaries in the identification of priority areas (Ch. 4). This approach responds to the call for ES research to focus more on the distribution of benefits and costs across different groups in society (Benton et al., 2018). We identified unique groups of ES beneficiaries based on their location and the characteristics of the ES flow, called ES flow zones. Our results showed that accounting for ES flow zones resulted in a more even distribution of ESs across these zones. Interestingly, our results also highlighted that a more even distribution of ESs did not necessarily reduce the overall conservation value for the whole of Europe. In other words, there is only a minor trade-off between protecting local and global needs for the conservation of ESs.

Previous studies highlighted strong trade-offs in conservation value based on local or global priorities for biodiversity. Coordinated identification of priority areas between countries for biodiversity is more effective than identification of priority areas for each country individually (Kark et al., 2009; Pouzols et al., 2014). Most likely, coordinated identification of priority areas for ESs is also most effective. Studies so far have presented methods with priorities either based on fully local or fully global preferences. Here we presented an approach that combined coordinated identification with local preferences. We identified priority areas for the whole of Europe (coordinated) whilst accounting for distribution of priority areas across each flow zone (local preference). For biodiversity and ESs alike, both general and location specific preferences for conservation are important to consider in conservation planning (Moilanen et al., 2013). Future research could test whether the method presented here can be used to minimize trade-offs between local and global preferences for conservation of biodiversity and ESs.

The method presented here is just a first step in integrating distributional issues into identification of priority areas. Previous, research presented methods to integrate issues of distribution conservation and restoration planning at the regional extent (Chan et al., 2006; Vollmer et al., 2016). We present a method that is able to account for beneficiaries in prioritization approaches across the spatial extent of the EU (Ch. 4). A technical limitation is the large amount of input files required and consequently the computational requirements. With increasing computational power and resources these limitations are manageable. However, the large spatial extent also requires a simplified conceptualization of beneficiaries. Here I discuss this limitations and venues for future research.

Spatial prioritization approaches require the delineation of beneficiaries based on spatial characteristics. We used flow zones as a spatial proxy for the distribution of beneficiaries. In reality, beneficiaries of ESs are, however, not a homogeneous group but use the environment in specific ways based on social cultural values for ESs. These sociocultural values are formed by both personal characteristics and social context (Scholte et al., 2015) . Therefore, one cannot really talk about a single beneficiary group, or a single demand for an ESs at a location. More likely, a variety of different groups of beneficiaries with unique preferences for a landscape exist at each location. This complicates and questions the use of spatial units as delineation for beneficiary groups.

Identifying sociocultural values is often not done in a spatial explicit manner and is difficult to perform at larger scales (Scholte et al., 2015). Recent progress is made in a study by Metzger et al. (2018) who used online surveys to determine demands from land in the EU for 2040. Distinct groups with unique demands for land use and the related benefits could be determined. This study partly resolved the issue that sociocultural values are difficult to determine at larger spatial scales. However, the results were not spatially explicit and can therefore not be easily used in spatial prioritization approaches.

A logical next step to integrate sociocultural values into spatial prioritization analyses is making sociocultural values spatially explicit across larger extents. If so, future work on identifying priority areas for ESs can integrate diverse demands for ESs as well as account for current and future demands for ESs. Distribution of ESs across beneficiaries can then be made not only on beneficiaries' location but also with regards to their sociocultural values for ESs and landscapes in general. An alternative is to develop different prioritization experiments, where each ES is valued according to the group characteristics. However, without the availability of these datasets differentiation between beneficiary groups for prioritization analysis will likely be based on spatial characteristics. The approach presented here provides a first simplified method to do so across larger extents.

7.4.3 Land use change and prioritization

We tested how changes in land use affect prioritizing areas for multiple ESs across the EU (Ch. 5). Specifically, we used scenarios to quantify the effects of changes in land cover, management intensity and configuration between 2000 and 2040 on the rank of priority areas and the level of ESs within priority areas. Land use change strongly affected the relative priority assigned to areas, resulting in large shifts in the priority area network between 2000-2040. Land use change had both positive and negative impacts on the ES quantified as well as on the priority rank of an area. We identified potential areas for restoration, conservation and stable priority areas.

Although priority areas shifted, the change in the level of ESs maintained within top priority areas was scale-dependent. For the whole of Europe, the changes in ESs over time within priority areas were limited. The capacity for nature-based tourism even increased over time. However, within Europe strong and regionally contrasting changes in ES capacity were observed. Often areas with increasing or decreasing ES capacity over time were spatially clustered. Although land use change can induce minor net changes in ESs at the European level, regional impacts can be strong and of significant importance to the people benefiting from these ESs.

The findings from chapter 5 question the common application of land use change in prioritization studies. Prioritization studies for biodiversity conservation generally consider land use change as a threat. This resulted in the use of negative uniform impacts of land use change on biodiversity, irrespective of the location or the species conserved. Studies on land use change and biodiversity or ESs commonly apply uniform negative effects (Luck et al., 2009; Pouzols et al., 2014; Cimon-Morin et al., 2016). Few prioritization studies actually assess the effect of land use change on ES capacity over time (but see Fan et al., 2016). Our results clearly showed that the impact of land use change differed between ESs and across space. In other words, depending on the location and ES studied impacts of land use change can range from strongly positive to strongly negative. This questions the use of uniform negative effects of land use change on ES capacity in prioritization studies.

Accounting for threat of land use change to priority areas can also inform land management. Land use change is an important driver of changes in ESs in Europe (Polce et al., 2016; Stürck and Verburg, 2017). Accounting for threats to priority areas, such as land use change, is important since it can further inform the management needs of the identified priority areas (Luck et al., 2012). A shortcoming of many prioritization studies is that they implicitly assume that identified areas will be conserved. We identified priority areas based on current land use. Some of these priorities were stable over time, or unthreatened by land use change, whereas others were negatively affected by land use change. Only the former require conservation management. Most interestingly, most priority areas did not face any threat of land use change, indicating that these areas do not necessarily require any active management intervention to maintain ESs over time. Similarly, we identified future priority areas that would benefit from active or passive restoration. These areas, and the level of ESs therein, would actually benefit from the projected land use change. Our approach to account for land use change in prioritization studies makes it possible to identify regions based on novel opportunities following land use change and avoid threats coming from anticipated land use change.

We found strong regional variation in the impact of land use change on ES capacity over time. Land use change impacted ESs in these regions and the beneficiaries within these regions differently. A major limitation is that we could only primarily focus on changes in ES capacity following land use change. However, to approximate ES demand we accounted for changes to ESs within areas of ES flow. To date, we are missing spatial projections of changes in ES demand. Ideally, ES demand and ES capacity are linked to estimates of over- and undersupply of an ES, so called ES budgets. However, the use of ES budgets is complicated by the fact that the quantities in which demand and capacity are expressed differ. One option is to use ES accounting, a way of expressing changes in ES demand and capacity, often in monetary terms (Bateman et al., 2013; Hein et al., 2016). ES accounting has also been used to estimate the impact of land use change on future ESs in the UK (Bateman et al., 2013). A limitation of ES accounting is the fact that it is often non-spatial and ignores regional differences. It thus ignores differing impacts of land use change on beneficiaries. Here we highlighted that impacts of land use change strongly differ between regions. For the beneficiaries in these regions, losses in ES capacity might not be offset by gains in ESs in other locations. Therefore, an important next step for ES accounting and ES research in general is to incorporate regional impacts of land use change on ESs and especially the local to regional impact on beneficiaries in the assessment methodology (Benton et al., 2018). This requires not only quantifying overall changes in ESs, but also requires identifying winners and losers of changes in ESs.

7.4.4 Management intensity and prioritization

Accounting for management intensity affected the relative priority ranking of areas in our study in the EU (Ch. 5). Management intensification, especially in forests, was a strong threat to the ESs pertained in priority areas identified under current land use. As a consequence, these high priority sites given current land use dropped in priority rank under the projected future conditions, as they no longer were expected to provide the highest value of ESs. Overall, management intensification was an important driver of shifts in priority areas. Therefore, future assessment should look beyond changes in land cover and include changes in management intensity.

Management intensity is not commonly incorporated in assessments of ES capacity and consequently in prioritization studies. Previous research already quantified the effect of cropland management on pollination (Zulian et al., 2013), changes in forest management over time on ESs and biodiversity (Verkerk et al., 2014) and changes in forest, cropland and grassland management on a number of ESs (Stürck and Verburg, 2017). Most prioritization studies however focus on a single assessment of current ES capacity without incorporating land use change (Adame et al., 2014; Casalegno et al., 2014; Cimon-Morin et al., 2014; Schröter et al., 2014b; Remme and Schröter, 2016). Our results highlight the importance of accounting for the effects that changes in management intensity and land cover can have on ES capacity and subsequently on site prioritisation.

7.4.5 Landscape configuration and prioritization

We incorporated effects of landscape configuration in the mapping of ESs (Ch. 5). Consequently, changes in land cover resulted in changes in ES capacity of the surrounding land through changes in the landscape configuration. However, effects of landscape configuration were not fully accounted for in chapter 5. Here I discuss some of the opportunities to further integrate configuration into prioritization analysis.

One option is to include connectivity requirements into prioritization analysis. In conservation studies researcher often prefer connected priority areas networks to many smaller areas (Schröter and Remme, 2016; Kukkala and Moilanen, 2017). However, connectivity requirements for ESs were not considered in the presented prioritization approach. Connectivity requirements would likely differ depending on the ES considered (Ch. 2). Nonetheless, a first step for ES research could be to assign a single connectivity requirement to all priority areas. Recently, techniques to implement connectivity requirements in reserve selection for ESs have been discussed, but so far these techniques have not been implemented (Kukkala and Moilanen, 2017). Other interesting recent work includes prioritizing areas in terms of climate connectivity (Carroll et al., 2018). These approaches can provide novel insights and methods for integrating connectivity requirements with ES prioritization.

However, for ESs larger connected areas are not necessarily preferred over smaller unconnected areas. We found that remaining fragments of natural vegetation and landscape elements within the matrix of urban and agricultural land uses obtained a high priority rank for ES (Ch. 4; Ch. 5). Pressures of land use change are strongest on these smaller remaining fragments. Managing smaller fragments in agricultural landscapes is both challenging and promising at the same time. Management of larger connected priority areas is viewed as relatively easier because of cost reductions and fewer threats of land use change on these areas (Luck et al., 2012). However, the importance of small patches for species and ecosystem functioning is often overlooked in conservation (Bodin et al., 2006; Tulloch et al., 2016; Wintle et al., 2019). These findings together with the work in this thesis highlight that prioritizing conservation of smaller fragments of natural vegetation can be crucial for biodiversity and sustain high level of benefits of ESs to human societies.

Therefore, maintaining small fragments and integrating connectivity requirements into prioritization for ESs might seem at odds. How to further integrate and balance connectivity requirements for ES capacity, whilst maintaining small patches of natural vegetation with high significance for ecological functioning and for meeting societal demands, is an interesting venue for further implementation of ESs into systematic conservation planning. Our results strongly suggest that small fragments of remaining habitat should be a main priority for conserving ESs. To do so maintaining ESs over time requires a landscape focus. This landscape focus should move beyond the conservation and management of single patches to include management of whole landscapes (Lovell and Johnston, 2009; O'Farrell and Anderson, 2010; Werling and Gratton, 2010; Turner et al., 2013; Grêt-Regamey et al., 2014; Mastrangelo et al., 2014; van Teeffelen et al., 2014; Benton et al., 2018). This landscape approach has to be tailored to simultaneously maintain small fragments with high societal ES value and larger connected areas of natural vegetation with high ecological functioning. Connectivity requirements for ESs are not uniform and have to be tailored to the ESs considered. We tentatively suggest that conserving global flow ESs might be best maintained within these larger connected areas whereas conserving local flow ESs requires maintaining smaller patches of natural habitat. The conservation of ESs thus needs to centre on balancing the needs of global and local flow ESs across the full gradient of diverse human-use landscapes.

7.4.6 From prioritization towards decision making to sustain ESs over time and space

We showed how spatial prioritization approaches can be used to identify priority areas for ESs. Zonation and related systematic conservation planning tools are very suitable to identify priority areas. An important question remains how selected priority areas should be managed. For biodiversity conservation the identification of priority areas is explicitly or implicitly assumed to result in the creation or expansion of protected area networks (Pouzols et al., 2014; Kukkala et al., 2016). Sometimes, priority areas for ESs are also linked to the establishment and expansion of protected area networks (Spanò et al., 2017). However, to obtain benefits from at least a set of ESs to society some form of minimal use of these areas is required. Therefore, management of priority areas for ESs should not necessarily focus on strict protection or conservation. The resulting prioritization should thus be interpreted as a hotspot map for ESs and not as a policy strategy for protected area establishment.

But how then should we manage priority areas for ESs? Additional analytical steps are necessary to determine management strategies for priority areas for ESs. One option is to explicitly consider different management alternatives in the prioritization of ESs (Wilson et al., 2006). An example of this is research in Norway where priority areas were identified for two levels of protection, i.e. strict protection and minimal use (Schröter et al., 2014b). Another option is to combine priority areas with threats of land use change (Ch. 5). Incorporating threats provides additional insights in the management required for these areas (Luck et al., 2012). Additional considerations for the identification and management of priority areas are costs of management and effectiveness of specific management measures (Naidoo et al., 2006; Wilson et al., 2006; Luck et al., 2012; Remme and Schröter, 2016). Much research has focused on assessing the sensitivity of priority area network for ESs to one of these conditions. Integrating all these aspects into a single prioritization analysis requires a major effort. However, it is a necessary step in bridging the gap between science and decision making on identifying management actions and priority areas to maintain ESs over time and space.

7.5 Restoring multifunctional landscapes

Combined changes in land cover, configuration and management intensity hold great potential for creating and restoring multifunctional landscapes with high ES capacity (Ch. 6). We used a landscape optimization tool to assess the potential of landscape restoration in a Dutch agricultural landscape. The landscape was restored for the objectives biodiversity, landscape aesthetics and fruit production at minimum costs to pasture farming. To balance the trade-offs between objectives, a combination of off-farm measures (taking land out of production) and on-farm measures (reducing management intensity and restoring linear landscape elements) proved most effective. We showed for the first time the potential to restore multifunctional landscapes by explicitly combining changes in land cover, configuration and management intensity.

Creating and restoring multifunctional landscapes is a crucial means to link human well-being and nature (Fischer et al., 2017b). Landscape optimization methods are especially suited to address questions on multifunctionality. These methods are designed to effectively balance multiple, often competing landscape objectives, showing the array of pareto-optimal outcomes. Landscape optimization analyses can highlight what could potentially be achieved through restoring landscapes and landscape elements (Nassauer and Opdam, 2008; Lovell and Johnston, 2009; Jones et al., 2013; Turner et al., 2013; Kremen and M'Gonigle, 2015). The identification of optimal trade-offs between objectives supports the creation of multifunctional landscapes (Cord et al., 2017; Fischer et al., 2017b).

Landscape optimization methods can also be used to compare the effectiveness of individual management options. Especially, it can provide insights in the effectiveness of a management option to balance multiple objectives. In chapter 6, we found that restoration of linear green elements can contribute to multiple objectives simultaneously. Linear green elements are an important component of the existing agricultural landscapes. The restoration of landscape elements can therefore be an important way of enhancing landscapes' capacity to provide ES (Jones et al., 2013). Interestingly, linear elements were capable of restoring multiple objectives simultaneously at low costs to pasture production. The results of Ch. 6 stress the relevance of looking beyond restoration of natural vegetation to explicitly consider management intensity and landscape configuration in identifying where and how to balance ESs and production within a single landscape.

7.5.1 Landscape optimization and decision making

Landscape optimization tools can provide relevant insights for decision making on land use planning and restoration measures. Our results were used in follow-up workshops with local decision makers and stakeholders. Here we discuss how to use results from optimization analysis in decision making on land restoration.

First, we showed that the existing nature restoration plan could be further improved by considering multiple objectives and costs simultaneously. Combining current restoration plans with landscape optimization analyses provides the opportunity to improve current plans and show the limits of what potentially could be achieved (Seppelt et al., 2013). This means that either the same objectives can be achieved at lower cost or for the same budget a higher level of objectives can be achieved.

Second, we tested the effectiveness of management options beyond taking land out of production. The current nature restoration plan for the area combines measures of taking land out of production, with voluntary restoration of agricultural land and the expansion of linear landscape elements on farmland (Provincie Utrecht, 2017). We showed that incorporating changes in management intensity of agricultural land and restoration of linear elements greatly improves the outcome of landscape restoration for all objectives considered, at lower costs to farmers. This result questions the current focus on taking land out of production over the other measures.

Third and last, we applied a recently developed method to identify priority areas for restoration actions. The current restoration plan allocated zones for restoration measures. However, the allocated zones showed little overlap with the outcomes of the optimization analysis and performed suboptimal on all objectives compared to the optimized outcomes. The landscape optimization analysis thus provided insight into what measures are needed to improve the current plans and where these measures are most effectively allocated.

However, there are also important challenges in using landscape optimization tools for decision making. A clear limitation of landscape optimization methods is the strong biophysical focus and the risk of technocracy. Mathematically optimal outcomes, as given by the landscape optimization approach, are not necessarily optimal or even desirable outcomes from a societal perspective. In democratic societies involvement of local stakeholders and decision makers is crucial for acceptance and putting plans for landscape change into practice (Opdam et al., 2013; Fürst et al., 2014). Including stakeholders in the planning an decision making process is therefore highly recommended (Albert et al., 2014b; Fürst et al., 2014; Sitas et al., 2014) and can partly overcome some of the issues related to practical applicability of the landscape optimization outcomes (Albert et al., 2014a). At the same time, involving stakeholders is not a panacea to achieve an societal optimal outcome (Albert et al., 2014a; Fürst et al., 2014). A first reasons is that facilitating stakeholder processes and involvement is often difficult (Casado-Arzuaga et al., 2013b; Hatton MacDonald et al., 2014; Liu and Opdam, 2014). Moreover, stakeholders need to have the required knowledge and need to look beyond short term gains and personal goals, with limited societal benefits (Cairns, 1996; Casado-Arzuaga et al., 2013a, 2013b; Fürst et al., 2014).

A greater potential lies in using landscape optimization tools as part of the decision making process together with stakeholders. Using these tools to test proposed solutions by stakeholders and to facilitate discussion can partly overcome the before mentioned issues. It can provide the necessary assessment framework to test the merits of proposals. Therefore, optimal outcomes from a landscape optimization procedure should be used to inform stakeholder processes and decision making, without prescribing optimal outcomes. Combining technical tools with local stakeholders needs and knowledge can together better inform decision making on how and where to restore landscapes for ESs.

7.6 Outlook – restoring and conserving our land for multiple, diverse ESs and beneficiaries

I studied how to improve ES assessment by accounting for landscape configuration and management intensity. Moreover, I studied how ESs can be integrated into spatial prioritization tools for maintaining and restoring ESs in Europe across space and time. In the prioritization analysis I explicitly incorporated effects of land cover, landscape configuration and management intensity and changes therein. This outlook focuses on some general conclusions that emerged across the individual chapters presented in this thesis.

In all chapters I studied effects of land management, landscape configuration and societal demand on multiple ESs. However, across the chapters the methodology on how to account for land management and landscape configuration differed. These differences in methodology were mostly driven by practical reasons such as data availability, the effect studied in relation to the resolution of the study or the requirements of tools applied in the study. For example, in chapter 4 we highlighted the importance of accounting for ES demand, but in chapter 5 we used a proxy for ES demand because no suitable data was available on future ES demand Similarly, we accounted for effects of configuration in mapping pollination in both chapter 2 and chapter 4. However, how I accounted for effects of configuration differed based on the resolution of the study (25x25m vs. 1x1km cells). This partly hampers drawing general conclusions across studies. Nonetheless, I did apply similar concepts related to incorporating management intensity and landscape configuration in all chapters. Therefore, based on the results of this thesis and broader insights from scientific literature I draw some overarching conclusions.

This thesis aims to improve the implementation of ESs into decision support tools to improve decision making on how to manage the land for multiple ESs. It thus has an explicit scientific but also an explicit policy support motivation. Throughout the thesis I have discussed how tools for identifying priority areas and actions can be applied for decision making. However, better tools and better scientific understanding do not necessarily result in better decision making. The use and uptake of any spatial decision support tool for land management depends on wider considerations such as the ease of use and the political process and timing (Primmer and Furman, 2012; Rosenthal et al., 2015; Bouwma et al., 2018). Therefore, better understanding of how to prioritize areas and actions for ESs does not have to result in better decision making. Many decisions on how to prioritize areas and actions, such as which ES to incorporate, how to value each service, whether or not to focus on demand or supply, are political in itself. Scientific studies and the application of these tools can make the consequences of these decisions explicit to policy and decision makers. To what extent this ultimately results in better decision is not the subject of this thesis and depends as much on scientific understanding as it depends on practical considerations or the political process. Nonetheless, general advice on decision making and developing EU policies for maintaining and conserving land for ESs can be drawn from this thesis. Below I represent the main scientific findings in relation to current and future EU policies.

The consideration of landscape configuration and management intensity is crucial for prioritizing areas and actions to conserve and restore ESs. Throughout this thesis I have shown how landscape configuration and management intensity affect the capacity of the landscape to provide ESs. I highlighted how landscape configuration affects ES capacity (Ch. 2) and how management intensity affects production and biodiversity (Ch. 3). Landscape configuration impacted a subset of ESs. Accounting for landscape configuration resulted in strong variation in ESs at the local scale. Local scale variation in ES capacity is crucial for prioritization studies, and effective identification of priority locations. I also showed that incorporating landscape configuration and management intensity has a strong impact on identifying priority areas and actions to conserve and restore land for multiple ESs. The configuration of ES demand and ES capacity areas and changes in management intensity strongly affected the prioritization of areas for multiple ESs across Europe (Ch. 4; Ch. 5). Moreover, both landscape configuration and management intensity were crucial for restoring ESs and biodiversity in an agricultural landscape (Ch. 6). Future research focused on prioritizing areas and actions for ESs should incorporate both landscape configuration and management intensity in their assessment to effectively map the spatial distribution of ESs and the changes over time. Moreover, incorporating landscape configuration and management intensity widens the toolbox for decision makers and landscape managers to effectively maintain and restore ESs in landscapes. Not accounting for these two facets of land means that we miss important threats and opportunities to conserve and restore land for multiple ESs over time in Europe.

By 2020, all member states need to map the current state of the ESs in their territory (European Commission, 2011). The primary focus has thus been on assessment of services and the development of indicators to support this assessment (Maes et al., 2012, 2016). However, indicators suitable to assess ESs are not necessarily suitable for prioritization (Verhagen et al., 2014). Post-2020 policies should focus on management interventions to maintain and restore ESs in landscapes in Europe. This is already partly adopted in policies on no-net-loss of biodiversity and ESs. Indicators used to assess ESs and changes therein should be sensitive to management interventions to assess the effectiveness, which include changes in the management and configuration of the land. Only then, prioritization approaches are capable of providing targeted support for policy development and decision making on where and how to maintain and restore ESs.

Societal demand for ESs strongly affects the identification of priority areas, especially for ESs with local and regional flow. Prioritizing for ESs requires accounting for both ES capacity and ES demand. For local and regional ESs the relative priority of sites might be more depended on the demand for a service than on the actual capacity of the landscape (Chan et al., 2006; Cimon-Morin et al., 2013; Ch. 4). There is limited spatial exchangeability for ESs with local and regional flow. This means that local losses in these ESs cannot be offset by distant gains in the same ES. This is especially relevant from the perspective of the people and society that obtain benefits from the respective services. Therefore, local and regional ESs need to be maintained across Europe. This is important for current as well as future generations. I showed that land use change has strong impacts on regional changes in ESs, meaning that land use change strongly impacts the beneficiaries' opportunity to obtain an ES. Prioritization approaches for ESs should base the identification of priority areas not only on what is prioritised (the service), but also by for whom we prioritise a particular ES and who is impacted by changes in land use over time. Any prioritization analysis for ESs should attempt to at least quantify winners and losers of proposed areas and actions. This means that research needs to quantify changes in ES and the impacts thereof not only at aggregate scales but also at the scale of the beneficiaries.

This result has important implications for no-net-loss policies. No net loss policies for biodiversity and ESs are increasingly being adopted by international and national governments, as well as corporations ([IFC] International Finance Corporation, 2012; Tucker et al., 2013; Rainey et al., 2015; Schulp et al., 2016; Cowie et al., 2018). These policies aim to achieve no net loss over time in biodiversity and ESs through avoiding, minimizing, restoring and offsetting impacts of land use change and other developments (Gardner et al., 2013; Cowie et al., 2018). Impacts of land use change on local and regional ESs means that to achieve no net loss, impacts, restoration and offsetting are sensitive to scale. Therefore, no net loss policies for ESs need not only focus on aggregate changes in ESs but also need to incorporate the diverse spatial scales at which ES benefits are being obtained. Otherwise, achieving no net loss at European or national scales runs the risk of creating (locally) undesirable outcomes. At a minimum, accounting for beneficiaries makes these potentially undesirable outcomes explicit to decision makers and stakeholders. Therefore, no net loss policies should be explicit at what scale no net loss is desired and address who is and is not benefiting from maintaining and restoring ESs.

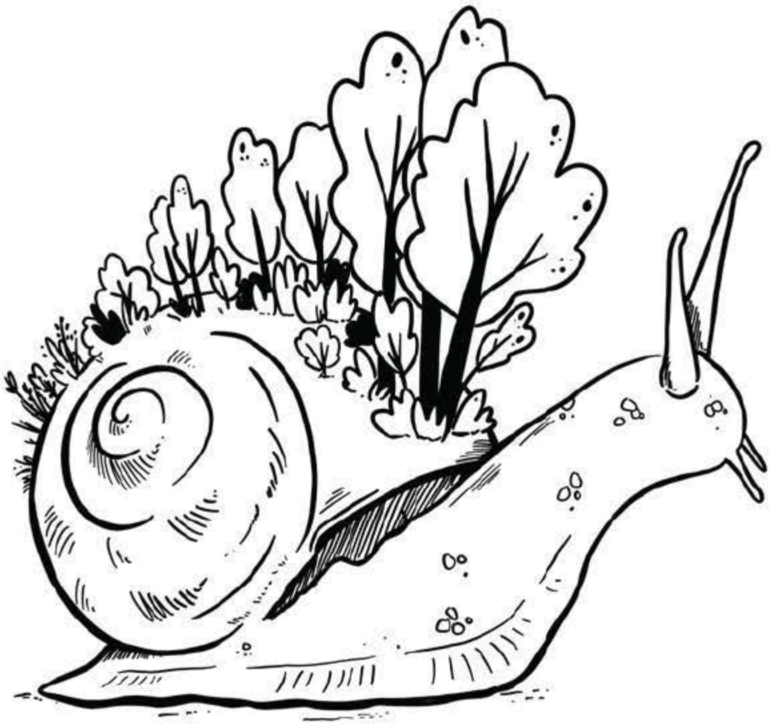
Conserving and restoring ESs requires a landscape approach. I showed the importance of landscape configuration for quantifying ES capacity (Ch. 2). I also showed the importance of configuration for prioritizing areas based on both ES capacity (Ch. 5) and ES flow (Ch. 4). Moreover, explicitly considering landscape configuration improved the restoration of ESs in an agricultural landscape (Ch. 6). Together these results highlighted that changes in ESs are affected by local and non-local conditions (Ch. 5). Explicitly considering configuration provides opportunities to design landscapes with improved benefits from ESs to society (Benton et al., 2018; Fischer et al., 2017b; Jones et al., 2013; Turner et al., 2013; Ch. 6).

I also found that small-scale landscape elements and fragments of natural vegetation are priorities for ES conservation and restoration (Ch.4; Ch. 5; Ch. 6). At the same time these fragments within the matrix of agricultural, urban and natural vegetation face increasing pressures from land use change (Ch. 5). Biodiversity and some services also benefit from relatively large unfragmented natural areas (Fahrig et al., 2011; Chaplin-Kramer et al., 2015). These diverse demands on landscapes need to be balanced, both within and across landscapes.

To better link human-well being and ESs, researchers and institutions have developed diverse concepts for a landscape approach. Landscape approaches hold the potential of meeting and balancing multiple needs. These approaches and concepts include multifunctional landscapes (Lovell and Johnston, 2009; O'Farrell and Anderson, 2010; Tscharntke et al., 2012; Fischer et al., 2017a, 2017b; Law et al., 2017; Stürck and Verburg, 2017), increasing landscape heterogeneity (Fahrig et al., 2011), forest and landscape restoration (Stanturl et al., 2012) or the need for sustainable intensification (Tilman et al., 2011; Garnett et al., 2013; Smith et al., 2016; Scherer et al., 2018). Although these concepts differ, the main aim is to realign production objectives with biodiversity and ESs in the human-nature matrix. To address these contrasting objectives, a landscape approach is most suitable and requires integrating human well-being with multiple, sometimes conflicting, ESs. This, however, does not mean that we need to solely manage land for multifunctional heterogeneous landscapes. Creating heterogeneous landscapes everywhere can have negative consequences for biodiversity as a whole (Fahrig et al., 2011) as well as for some ESs (Chaplin-Kramer et al., 2015). Moreover, designing multifunctional landscapes with lowered food production might result in negative externalities through increasing pressures on other landscapes and the related biodiversity, ESs and human well-being (Phalan et al., 2011b; Verburg et al., 2016). Planning conservation and restoration of ESs need to be adapted to the needs of local stakeholders, but be placed within requirements for achieving global goals on biodiversity conservation, carbon sequestration and food production. Therefore, multifunctionality of landscapes and their resulting services needs to be strived for both within and across landscapes.

Current policies on green infrastructure in Europe might be particularly suited to address the above issues (European Commission, 2013; Maes et al., 2015a). Green infrastructure creates a network of smaller landscape elements and fragments within and across landscapes used by humans. Hence green infrastructure supports ESs capacity and flow in landscapes used by humans. The ultimate goal is to maintain and improve the delivery of ES to society across Europe. Policies on green infrastructure thus need to place the priorities of local ES flow within European wide priorities for maintaining ES capacity and flow of a diverse set of ESs.

Together, the topics in this thesis address how to identify priority areas and priority actions to manage the land for a diverse set of ESs. We showed that societal demand for ESs shifts priority locations from remote natural areas towards smaller remnants of natural vegetation within the matrix of urban and agricultural land use. The resulting priority areas in agricultural and urban landscapes provide high benefits from nature to society. Managing these priority areas is challenging because threats of land use change are most severe in these areas. Moreover, threats are manifold consisting of changes in land use, configuration and management intensity. At the same time, it is also in these landscapes, and through combined management of changes in land use, configuration and management intensity, that the greatest opportunities lie to restore multifunctional landscapes.



8. Appendix

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8.2 Supplementary material

Chapter 2

S1: Additional tables and figures, including search terms for literature review.
https://static-content.springer.com/esm/art%3A10.1007%2Fs10980-016-0345-2/MediaObjects/10980_2016_345_MOESM1_ESM.pdf
S2: ES models used to calculate effect of composition and configuration on ES capacity
https://static-content.springer.com/esm/art%3A10.1007%2Fs10980-016-0345-2/MediaObjects/10980_2016_345_MOESM2_ESM.pdf

Chapter 3

All Supplementary material can be found online directly with the article:
https://doi.org/10.1111/gcb.14606
S3: Figure - PRISMA diagram.
S4: Full web of science search term.
S5: Characterizing land-use intensity classes.
S6: Description of datasets used in the analysis.
S7: Figure - Distribution of screened studies containing information on land use, biodiversity and yield.
S8: Five major stages of land-use history applied in the analysis.
S9: Pairwise comparison of predicted response ratios for land-use history and climate.
S10: Number of samples and percentage change in species richness and yield.
S11: Figure – Analysis of land use history as an explanatory factor.
S12: Figure – Analysis of climate as an explanatory factor.
S13: Figure – Analysis of potentially correlated or confounded variables.
S14: Figure – Effects of yield change with respect to nutritional value.
S15: List of references used in the meta-analysis.

Chapter 4

S16: Method to map urban leisure.
https://www.readcube.com/articles/supplement?doi=10.1111%2Fcobi.12872&index=1&ssl=1&st=bcc224baca17d6dbb9013503ff28fe5d&preview=1
S17: Zonation settings and scripts.
https://www.readcube.com/articles/supplement?doi=10.1111%2Fcobi.12872&index=2&ssl=1&st=bcc224baca17d6dbb9013503ff28fe5d&preview=1
S18: Additional Figures and Tables.
https://www.readcube.com/articles/supplement?doi=10.1111%2Fcobi.12872&index=0&ssl=1&st=bcc224baca17d6dbb9013503ff28fe5d&preview=1

Chapter 5

All supplementary material can be found online directly with the article:
https://www.sciencedirect.com/science/article/pii/S1470160X18300190
S19: Figures- Additional figures
S20: Zonation settings for the analysis.

Chapter 6

All supplementary material can be found online directly with the article:
S21: Figures - Land use Land Cover Map and 3D Visualization of optimization outcomes.
https://ars.els-cdn.com/content/image/1-s2.0-S1462901117310018-mmc3.docx
S22: Methods - Quantifying the environmental objectives.
https://ars.els-cdn.com/content/image/1-s2.0-S1462901117310018-mmc1.docx
S23: Methods - Settings for the optimization tool, CoMOLA.
https://ars.els-cdn.com/content/image/1-s2.0-S1462901117310018-mmc2.docx

8.3 Summary

Many landscapes are managed for agricultural and forestry products resulting in the unintended decline of cultural and regulating ecosystem services (ESs). These regulating and cultural ESs provide benefits to human society such as carbon sequestration, flood regulation, recreational opportunities or pollination of crops. In the future, rising populations and wealth are expected to further increase pressures on land and associated ESs. To maintain a diverse set of ESs over time, several policies aim to protect and restore land for ESs.

However, funds and land to maintain ESs are limited. Therefore, it is important to allocate the available budgets as effective as possible. In other words, it is important to identify priorities for land management both in terms of where (location) and how (actions) ESs should be maintained. Researchers have developed tools to prioritize areas and actions. These tools were mostly developed for biodiversity conservation and are now also used to identify priorities for ESs. The topic of this thesis is how to integrate ESs in the identification of priority areas and actions to maintain ESs over time.

The first part of this thesis focuses on improving the assessment of ESs. Prioritization approaches rely on maps of ESs as an input. Much effort has been put into quantifying the spatial variation, or mapping, of ESs. Land cover types are often used as proxies for ES levels. However, ESs levels can also depend on the spatial arrangement of land cover (landscape configuration). Moreover, ESs levels can depend on the intensity with which the land is being managed (management intensity). Whereas much effort has gone into linking land cover to ESs, we still have limited knowledge on how landscape configuration and management intensity affect the spatial variation in ESs.

In the first study of this thesis (Ch. 2) we focused on how landscape configuration affects ES capacity and the consequences thereof for mapping ESs. We reviewed the existing literature on effects of landscape configuration on ten ESs in the temperate zone. Only for a subset of ESs there is enough evidence to conclude that landscape configuration affects these ESs. More importantly, how and what aspects of landscape configuration affect an ES depends on the ES studied. We used the findings from this review to account for landscape configuration in the mapping of ESs. We mapped four ESs in Scotland, once with and once without accounting for landscape configuration. Accounting for landscape configuration results in strong variation in ES capacity at the local scale, for all four ESs considered. Across larger extents these local differences tend to average out and landscape configuration hardly impacts the overall level of ESs. The strong effect of landscape configuration on local variation in ES capacity is especially relevant when identifying priority areas for ESs.

In the second study (Ch. 3) we performed a meta-analysis on the simultaneous effects of management intensification on biodiversity and production. Whereas several studies perform a meta-analysis on management intensity and either biodiversity or production, this is the first study that addressed the response of both of these factors. We studied management intensification in agricultural and forestry systems. We conclude that there is a general trade-off following management intensification resulting in a decline in biodiversity and an increase in production. This trade-off is strongest in agricultural systems compared to grassland or forestry systems and for plant compared to animal species. Interestingly, the effect of management intensification on biodiversity and production was governed by the initial starting intensity and the extent of intensification. How management intensity shapes the trade-off between production and biodiversity is a topic of great interest for global sustainability.

The first two studies centred on improving the assessment of ESs. In the latter chapters we studied how ESs can be integrated and applied in spatial prioritization studies. Prioritization studies aim to identify where and how to best manage the land for a set of objectives (here ESs).

Humans obtain diverse benefits from nature. These benefits are not only shaped by the capacity of a landscape to provide an ES, but also by the human demand for an ES. Thus, ESs are for a large part shaped by the societal demand. This notion of societal demand is used in Chapter 4. In Chapter 4 we identified priority areas for ESs at the extent of the European Union (EU), based on maps for five ESs. We compared priority areas for ES capacity and ES flow, the combination of ES capacity and ES demand. There are strong differences in the identified priority areas with and without accounting for ES demand. Identifying priority areas for ES capacity resulted in locations with good ecological functioning but with limited benefits to human society. Identifying priority areas for ES flow shifted locations towards patches of natural vegetation close to cities and agricultural land, with increased benefits to human society. In a second comparison we accounted for spatial restrictions to ES flow. ES flows differ between individual ESs, ranging from strictly local to fully global. Therefore, the opportunity to actually benefit from an ES is spatially restricted by the ES flow. As an example, a priority area for flood regulation in Spain (Ebro basin) does not contribute to maintained flood regulation in Poland (Oder basin). We developed a method to account for the spatial flow of each ES and the distribution of priority areas across individual flow zones. Our results show that accounting for spatial flow of each ES results in a better distribution of maintained ESs across regions in the EU with only a minimal impact on the overall level of ESs maintained. This research highlights the opportunity to maintain ESs for the entire EU and across regions in the EU.

A limitation of Chapter 4 is that we identified priorities only for current land use. Land use change is one of the most important drivers of change in ESs across Europe. Moreover, land use change can inform the management of priority areas. Therefore, in Chapter 5 we identified priority areas for ESs in 2000 and in 2040 in the European Union (EU) for a total of five ESs. We used two scenarios on land use change that mainly differ in the extent of agricultural land abandonment. We accounted for changes in land cover, landscape configuration and management intensity and quantified the impact thereof on ESs and the location of priority areas. Management intensity is a strong driver of changes in priority areas for ESs. We identify the most important drivers of change in ES capacity over time in priority areas and can therewith distinguish between areas most suitable for conservation and areas most suitable for restoration. Accounting for land use change results in a strong shift in the location of priority areas over time. Although locations of priority areas shift between 2000 and 2040, the effect of land use change on overall ES capacity for the entire EU is limited. However, there is strong regional variation in the effects of land use change on ES capacity. Existing policies aim for a no-net-loss of ESs at the level of the EU. To maintain ESs across the entire EU, policies should aim for no-net-loss of ESs at EU level and at regional level.

Chapters 4 and 5 have mostly focused on identifying priority areas for ESs, often implicitly assuming that these areas will be conserved. However, in reality there are multiple alternative management options (actions). In the last chapter (Ch. 6) we study priority actions for the restoration of ESs in agricultural landscapes (Kromme Rijn). Specifically, we conducted a case study in a Dutch agricultural landscape. Here, we compared actions to restore land and the resulting ESs through 1) restoration of natural vegetation on agricultural land taken out of production, 2) restoration of linear landscape elements on agricultural land or 3) changing management of agricultural land. We optimized the landscape for three objectives, namely crop production, recreational opportunities and biodiversity, at a minimum loss to grassland production. The outcomes of our study were compared to the existing nature management plan for the area. We show that for either the same loss in grassland production more area can be restored or that the same outcomes can be achieved for lower losses in grassland production. To minimize trade-offs between individual objectives requires combining all three restoration actions, otherwise strong trade-offs between objectives occur. We also show that the currently identified priority areas in the nature management plan for restoration of natural vegetation may be suboptimal. This research highlights the opportunity and the practical relevance of restoring landscapes for multiple ESs through changes in land cover, landscape configuration and management intensity.

Together, the topics in this thesis address how to identify priority areas and priority actions to manage the land for a diverse set of ESs. We show that societal demand for ESs shifts priority locations from remote natural areas towards smaller remnants of natural vegetation within the matrix of urban and agricultural land use. The resulting priority areas in agricultural and urban landscapes provide high benefits from nature to society. Managing these priority areas is challenging because threats of land use change are most severe in these areas. Moreover, threats are manifold consisting of changes in land use, configuration and management intensity. At the same time, it is also in these landscapes, and through combined management of changes in land use, configuration and management intensity, that the greatest opportunities lie to restore multifunctional landscapes.

8.4 Dankwoord

A regular PhD ceremony ends with a laudatio for the person graduating. However, obtaining a PhD degree is impossible without the help and support of many other people. Therefore, I would like to thank everyone who made a contribution. I have been lucky enough to meet colleagues and new friends over the past years. Some were inspiring, some were supporting, some were kind, and some were there to celebrate your achievements or to support you during tough times. But most importantly, I was lucky enough to be surrounded by nice and fun people to drink coffee with, eat cookies, read books, go to dinner and parties, play board games, eat ice cream with or go for pizza on Friday.

First and foremost I would thank my supervisors Peter Verburg and Astrid van Teeffelen. They have literally been there from day one. Peter, you taught me what it requires to be a scientist. I genuinely enjoyed our discussions on the research papers and your strive to always deliver good work. Thanks to you I was also able to write a book chapter and to participate in the LUBDES workshops. For the rest, I'll try to keep the writing in this laudatio "to the point". Astrid, I honestly do not think that there is a single sentence of my work that you have not read and commented on. But you always did so with a smile and a positive note. You're a wonderful person and supervisor to work with. I have not yet found a new colleague with the "unhealthy" habit of giving me Tony's chocolate bars to celebrate achievements or to cheer me up when needed.

I would also really like to thank some of my co-authors. I enjoyed working with a diverse and international group of scholars. Alessandro, it was great to visit you in Aberdeen. You taught me a lot about Scotland, about biophysical processes and the mapping of ESs. I hope we keep meeting each other at conferences like Ghent. Aija, you were instrumental to the work done in the third chapter. I had zero experience in performing Zonation runs but had the crazy idea to do six experiments with several 100.000 input files. You made this crazy idea into reality and kept your calm when inevitably runs had to be done over and over again. Emma, I really enjoyed working with you on the last chapter. Most of the time work as a PhD is quite lonely, but this project really felt like a shared one. I enjoyed our meetings in which we discussed our current work and our future work and plans.

Not everyone who makes a significant contribution becomes a co-author. I would like to thank David, Nynke and Joona for all the help provided throughout the years. Both David and Nynke helped me a lot with the first paper. David, thanks for all your help, both in terms of practical and scientific issues and in terms of swimming condition. Nynke, much of the work I have done throughout my thesis is based on your previous work. I could not have done so without you. Joona, thank you for all the help on my third paper. I'll definitely remember all the great meetings together with Astrid on prioritization work for biodiversity and ESs. The wine sure did help our discussions flow.

The LUBDES project was a special part of my PhD. It was great to meet everyone in Leipzig and Annapolis for several workshops. Days were long and we worked hard. We read, coded, recoded and re-recoded way to many papers and had an incredible amount of conference calls. But each of us was looking forward to the next workshop, to work on the project and see each other again. The work done really feels like a team effort and I'm grateful for the opportunity to meet you all. Each Friday I still have the habit of logging in for a conference call.

But of course the core group of colleagues consisted of EG. The EG team is great. Throughout the years, many PhDs, visiting researchers and post-docs come and go. There has always been a nice social vibe with all the colleagues. Together, we make it a nice place to work. I'll remember the cookie box, EG meetings, journal clubs and pizza Friday. A special note to everyone from the book club where we learned that we can discuss and enjoy more than scientific literature.

I spent my time as a PhD in two offices. When I started I shared an office with Marthe and Julia. All three of us had their unique working rhythms. Some of us came in early, some of us came in late (because they first had to do laundry). But when we were all present our office was filled with life, lots of plants, peanut butter cans and fun. The last years I shared an office with Reinhard and Ziga. Every day started with the Ziga fun fact of the day. We were probably the most visited and popular office, partly due to us and partly due to people who were looking for the cookie box or a stapler.

A regular PhD ceremony is with two paranymphs, but I sort of have three. First Katharina, the *not to be* paranymph. As you say yourself, you always come up with an excuse to not join important events in my life. I really enjoyed having you as a colleague. We had great talks about life over coffee, beer and books. I often miss your humour. I would also like to thank my official two paranymphs; Ziga and Jurre. Jurre, we did the same bachelor and master degree. When we graduated we celebrated our master degree together. Throughout my studies you have been there as a friend and study mate. Ziga, you are a great colleague and a really fun office mate. I still have your picture on my office desk. You made everyday work fun and enjoyable. I'm glad and grateful to be standing on the podium for my PhD ceremony with the two of you.

Last, I would like to thank Fenne. We started our PhDs on exactly the same day and roughly finish around the same time. We both work with a mouse and use a computer but that's probably the end of the similarities between our research fields. At home, we can talk about our work, supervisors, reviewers and much more. But most importantly you can make me forgot about work and enjoy all other good things in life.

Thanks to everyone who made a contribution to the thesis and to everyone who made these past years into a memorable and joyful time,

Willem

8.5 SENSE research school diploma

Netherlands Research School for the
Socio-Economic and Natural Sciences of the Environment

DIPLOMA

For specialised PhD training

The Netherlands Research School for the
Socio-Economic and Natural Sciences of the Environment
(SENSE) declares that

Willem Verhagen

born on 17 May 1988 in Helmond, The Netherlands

has successfully fulfilled all requirements of the
Educational Programme of SENSE.

Amsterdam, 14 June 2019

The Chairman of the SENSE board	the SENSE Director of Education
Prof. dr. Martin Wassen	Dr. Ad van Dommelen

The SENSE Research School has been accredited by the Royal Netherlands Academy of Arts and Sciences (KNAW)

KONINKLIJKE NEDERLANDSE
AKADEMIE VAN WETENSCHAPPEN

The SENSE Research School declares that **Willem Verhagen** has successfully fulfilled all requirements of the Educational PhD Programme of SENSE with a work load of 41.2 EC, including the following activities:

SENSE PhD Courses

- Environmental research in context (2013)
- Research in context activity: 'Writing book chapter titled 'Mapping ecosystem services' (in book 'Ecosystem services - from practice to concept') and using this chapter as course material. Co-organizing Journal Club of Environmental Geography group (2016-2017)'

Other PhD and Advanced MSc Courses

- Introduction to R for statistical analysis, Wageningen Graduate Schools (2014)
- Techniques for writing and presenting a scientific paper, Wageningen Graduate Schools (2014)
- Philosophy of science, Vrije Universiteit Amsterdam (2014)

External training at a foreign research institute

- SESYNC: funding for participation in synthesis workshops on the relation between land management, production and biodiversity, Annapolis, USA & sDiv, Leipzig, Germany (2014 - 2016)

Management and Didactic Skills Training

- Supervising two MSc students with thesis entitled 'Quantifying the effects of woody elements on ecosystem service supply and their potential to balance ecosystem service trade-offs in agricultural landscapes'(2014) & 'Reviewing the impact of land use intensification on species abundance'(2017)
- Teaching in the BSc field course 'Limburg' for Earth and Economics students (2016;2017)

Oral Presentations

- *OPERAs: Scotland as a joint study areas to enhance ES quantification, valuation and governance.* EsCom conference, 28-29 April 2014, Edinburgh, Scotland
- *Shifting priorities for ecosystem services under land use change.* IALE conference: From pattern and process to people and action, 12-16 September 2017, Ghent, Belgium

SENSE Coordinator PhD Education

Dr. Peter Vermeulen

8.6 Publication List

Peer-reviewed Publications

Beckmann, M., Gerstner, K., Akin-Fajiye, M., Ceauşu, S., Kambach, S., Kinlock, N.L., Phillips, H.R.P., **Verhagen, W.,** Gurevitch, J., Klotz, S., Newbold, T., Verburg, P.H., Winter, M., Seppelt, R., 2019. Conventional land-use intensification reduces species richness and increases production: A global meta-analysis. Glob. Chang. Biol.. doi: 10.1111/gcb.14606

Lautenbach, S., Mupepele, A.C., Dormann, C.F., Lee, H., Schmidt, S., Scholte, S.S.K., Seppelt, R., van Teeffelen, A.J.A., **Verhagen, W.,** Volk, M., 2019. Blind spots in ecosystem services research and challenges for implementation. Reg. Environ. Chang. 1–22. doi:10.1007/s10113-018-1457-9

Verhagen, W., van der Zanden, E.H., Strauch, M., van Teeffelen, A.J.A., Verburg, P.H., 2018a. Optimizing the allocation of agri-environment measures to navigate the trade-offs between ecosystem services, biodiversity and agricultural production. Environ. Sci. Policy 84. 186-196. doi:10.1016/j.envsci.2018.03.013

Verhagen, W., van Teeffelen, A.J.A., Verburg, P.H., 2018b. Shifting spatial priorities for ecosystem services in Europe following land use change. Ecol. Indic. 89. 397-410. doi:10.1016/j.ecolind.2018.01.019

Verhagen, W., Kukkala, A.S., Moilanen, A., van Teeffelen, A.J.A., Verburg, P.H., 2017. Use of demand for and spatial flow of ecosystem services to identify priority areas. Conserv. Biol. 31, 860–871. doi:10.1111/cobi.12872

Seppelt, R., Beckmann, M., Ceausu, S., Cord, A.F., Gerstner, K., Gurevitch, J., Kambach, S., Klotz, S., Mendenhall, C., Phillips, H.R.P., Powell, K., Verburg, P.H., **Verhagen, W.,** Winter, M., Newbold, T., 2016. Harmonizing Biodiversity Conservation and Productivity in the Context of Increasing Demands on Landscapes. Bioscience 66, 890–896. doi:10.1093/biosci/biw004

Verhagen, W., Van Teeffelen, A.J.A., Baggio Compagnucci, A., Poggio, L., Gimona, A., Verburg, P.H., 2016. Effects of landscape configuration on mapping ecosystem service capacity: a review of evidence and a case study in Scotland. Landsc. Ecol. 31, 1457–1479. doi:10.1007/s10980-016-0345-2

Non-peer reviewed publications

Seppelt, R., Beckmann, M., Ceauşu, S., Cord, A.F., Gerstner, K., Gurevitch, J., Kambach, S., Klotz, S., Mendenhall, C., Phillips, H.R.P., Powell, K., Verburg, P.H., **Verhagen, W.,** Winter, M., Newbold, T., 2019. Trade-Offs and Synergies Between Biodiversity Conservation and Productivity in the Context of Increasing Demands on Landscapes, in: Atlas of Ecosystem Services. doi:10.1007/978-3-319-96229-0_39

Schaller, L.L., Targetti, S., Kantelhardt, J., Zavalloni, M., D'Alberto, R., Raggi, M., Nikolov, D., Boevsky, I., Borisov, P., Radev, T., Anastasova, M., Bareille, F., Dupraz, P., Byg, A., Faccioli, M., Kyle, C., Roberts, M., Mantymaa, E., Juutinen, A., Tyrvainen, L., Kurttila, M., Gomez-Limon, J.A., Gutierrez-Martin, C., Villanueva, A.J., Castillo, M., Berbel, J., Ratinger, T., Vancurova, I., Bavorova, M., Havova, R., Kespaik, A., Lassur, S., Tafel-Viia, K., Kuttim, M., Mihai, C., Maxim, A., Apostoaie, M.C., Letki, N., Czajkowski, M., Zagorska, K., Komossa, F., **Verhagen, W.,** van der Zanden, E.H., Verburg, P.H., Hafner, K., Zasada, I., Piorr, A., Nanett Trau, F., Viaggi, D., 2018. Deliverable 5.2: Report on comparative evaluation results. Vienna, Austria.

Verhagen, W., Stürck, J., Schulp, C.J.E., Verburg, P.H., 2014. Mapping Ecosystem Services, in: van Beukering, P.J.H., Bouma, J. (Eds.), Ecosystem Services: From Concept to Practice. Cambridge University Press, Cambridge (UK), p. 22

8.7 About the author

Willem Verhagen is an environmental scientist. He is most interested to work on interdisciplinary topics that connect human society with global environmental sustainability. Willem obtained a bachelor degree in Earth and Economics from Vrije Universiteit Amsterdam (2010) and a masters degree, cum laude, in Sustainable development from Utrecht University (2013). After his masters, Willem worked as a PhD student in the Environmental Geography group under the supervision of prof. P.H. Verburg and dr. A.J.A. van Teeffelen. His research focuses on identifying priority areas and actions to maintain ESs in Europe. During his PhD Willem obtained funding from the socio-environmental synthesis centre (SESYNC) and sDiv (Leipzig) for a project on synthesizing the current knowledge on land use, biodiversity and ecosystem services (LUBDES project). His research is published in international journals, book chapters and was presented at several conferences. Besides his research, Willem has supervised master students and taught courses for Bachelor students Since November 2017, Willem works as a policy researcher at the Netherlands Environmental Assessment Agency (PBL). His current work focuses on quantifying current and future impacts of land degradation and the opportunities to resolve these impacts through sustainable land management and restoration. Besides his work Willem loves to go swim outdoors, go hiking or to read, discuss and write fiction. He therefore hopes that you enjoy reading this thesis.

Want to keep up to date with recent work, please visit:
https://www.researchgate.net/profile/Willem_Verhagen
https://www.linkedin.com/in/willemverhagen/

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The research leading to the results presented in this thesis received support from the OPERAs project from the European Union's Seventh Framework Programme under grant agreement number 308393.

Printing of this thesis was financially supported by the Faculty of Earth and Life Sciences, VU University Amsterdam.
Illustrations: Aafke Mertens. www.curlysketches.nl
Printed by: Print Service Ede, The Netherlands
ISBN: 978-94-92679-90-1